UNIVERSITÄT FÜR BODENKULTUR WIEN
University of Natural Resources
and Life Sciences, Vienna

Master Thesis

Phosphorus adsorption capacities of stream sediments in Lower Austria with relation to land use and riparian vegetation

submitted by

Augustinus LIECHTENSTEIN, BSc

in the framework of the Master programme

Umwelt- und Bioressourcenmanagement

in partial fulfilment of the requirements for the academic degree

Diplom-Ingenieur

Vienna, September 2022

Supervisor:

Ass.Prof. Priv.-Doz. Mag. Dr. Gabriele Weigelhofer
Institute of Hydrobiology and Aquatic Ecosystem Management
Department of Water, Atmosphere and Environment

Abstract

Streams with surrounding agricultural areas are exposed to large nutrient loads that enter through runoff and erosion. These diffuse nutrient sources are particularly difficult to mitigate and have become a major issue in stream management. The consequences are severe for both the stream and recipient water bodies. Riparian buffer strips are a measure which increases the retention of nutrients and soil particles in the terrestrial zone and increases the ability of the stream sediment to adsorb and thereby retain nutrients. This study has investigated the effects of riparian buffer strips on phosphorus retention by benthic sediments. A comparison of adsorption characteristics of eight streams, each having a non-riparian-vegetated and a riparian-vegetated site, was performed. Specifically, six-batch equilibrium experiments were applied to calculate the equilibrium phosphorus concentration (EPC_0). In addition, sediment respiration, sediment grain sizes, organic matter, sediment inorganic phosphorus and water nutrients were sampled to obtain insights into bacterial activity and sediment qualities. The hypothesized larger retention capacities at riparian-vegetated sites could not be confirmed. Differentiation of the results by land use was possible, as half of the streams were surrounded by cropland and half by pasture. Due to the increased runoff and erosion in cropland, positive effects of riparian vegetation on the retention capacity of the sediment were expected and demonstrated to be present. In this study, streams surrounded by pasture did not exhibit increased sediment buffer capacities due to the riparian vegetation; however, the results from cropland streams indicate that riparian buffers are of particular value in intensive agriculture.

Keywords: Equilibrium phosphorus concentration (EPC_0), riparian buffer strips, agricultural streams

Zusammenfassung

Bäche in landwirtschaftlicher Landnutzung sind großen Nährstoffmengen ausgesetzt, welche durch Erosion und Abfluss eingetragen werden. Diese diffusen Nährstoffquellen sind schwer zu bewältigen und stellen ein zentrales Problem für die Ökologie der Bäche dar. Die Konsequenzen sind schwerwiegend und betreffen sowohl den Bach als auch nachfolgende Gewässer. Ufergehölz longitudinal zum Bach ist eine Maßnahme welche zunächst Nährstoffe und Bodenpartikel im terrestrischen Bereich zurückhält und des Weiteren die Fähigkeit von Sedimenten, Nährstoffe zu adsorbieren, steigert. Diese Studie beabsichtigt die Effekte, welche Ufergehölze auf das Phosphor Retentionspotenzial von benthischen Sedimenten hat, zu untersuchen. Dafür wurden die Phosphor Adsorptionscharacteristiken von Standorten mit und ohne Ufergehölzen von 8 verschiedenen Bächen verglichen. Es wurden sechsreihige Gleichgewichts Experimente unternommen um Phosphor Gleichgewichtskonzentrationen (EPC_0) der Sedimente zu eruieren und damit deren Phosphor Adsorptionspotenzial. Zusätzlich wurde die Sediment Respiration, die Sediment Korngrößen, der organische Gehalt, der anorganische Phosphor im Sediment und die in der Wassersäule befindlichen Nährstoffe untersucht um mikrobiologische Aktivitäten und Sediment Qualitäten zu erhalten. Die Resultate konnten die Annahme von größeren Retentionspotenzialen für Standorte mit Ufergehölz nicht bestätigen. Eine Differenzierung der Resultate nach Landnutzung war möglich, da sich die Hälfte der Bäche in Weidefläche und die andere in Ackerfläche befunden hat. Da Ackerfläche auf mehr Erosion und Abfluss exponiert ist wurde eine bessere Effektivität der Ufergehölze in Ackerland Bächen vermutet und durch die Ergebnisse bestätigt. In dieser Studie wurde also bei Weideland umgebenen Bächen keine erhöhte Wirkung von Ufergehölzen auf die Nährstoffretention von Bachsedimenten nachgewiesen. Jedoch ist klar abgezeichnet, dass Ufergehölze, in Bächen mit intensiver Landwirtschaftlicher Nutzung, von besonderem Wert für die Mitigation von Nährstoff Einflüssen sind.

Schlüsselwörter: Phosphor Gleichgewichtskonzentrationen (EPC_0), Ufergehölze, Landwirtschaftliche Bäche

Acknowledgements

I would like to thank my supervisor, Ass.Prof. Priv.-Doz. Mag. Dr. Gabriele Weigelhofer, for giving me the opportunity of this master's thesis and providing me with the guidance I needed. It was a great pleasure to conduct this experimental research at the WasserCluster Lunz, and it was highly interesting and exciting to join the research group BIGER (Biogeochemistry and Ecohydrology of Riverine Landscapes). I want to thank the entire group for including and helping me in many ways. Especially, I want to thank Elmira Akbari, Beate Pitzl, Annette Puritscher and Irina Ludwig for their help and support in the laboratory work.

Contents

List of Tables

List of Figures

List of Abbreviations

EPO_0	Equilibrium phosphorus concentration
SRP	Soluble reactive phosphorus
P_{sorp}	Amount of desorbed or adsorbed SRP
SRP_{ini}	Initial SRP concentration of the water
SRP_{fin}	Final SRP concentration of the water
DOC	Dissolved organic carbon
V	Volume
W_{sed}	Dry weight of the sediment
W	Watt
g	Gram
M	Mol
rpm	Rounds per minute
mg P kg^{-1}	Milligram phosphorus per kilogram
µg P L^{-1}	Microgram phosphorus per litre
t ha^{-1} y^{-1}	Ton per hectare per year
mg g^{-1} h^{-1}	Milligram per gram per hour

1. Introduction

Continuously increasing nutrient loads in freshwater systems have become a major issue for the management of streams (Stutter et al., 2018). Consequences can be severe for both the stream and recipient water bodies. In addition, the management of diffuse nutrient inputs has become particularly complicated (Weigelhofer et al., 2018). This study has investigated the effects that riparian vegetation can have on the ability of a stream to retain phosphorus and thus mitigate the effects of phosphorus inputs in agricultural streams.

1.1 Eutrophication

The growth and life of stream (micro-) organisms are sustained by nutrients. However, excessive loads of nutrients can drastically affect aquatic systems. The appearance of increasing amounts of nutrients in freshwater systems is referred to as eutrophication (Stutter and Lumsdon, 2008; Woodward et al., 2012). The cause of eutrophication is nutrient overloading through increased inputs. This can be induced by land use changes, as these changes can decrease the retention of nutrients and increase nutrient input into aquatic systems. Modifications of the hydrological cycle can further decrease the potential of streams to retain nutrients (Jarvie, 2013). Among the consequences are excessive growth of algae and macrophytes, the reduction of macrophyte species, oxygen depletion and large changes in pH values (Edmondson and Vollenweider, 1968; Schindler, 2006). In Europe, thousands of freshwater systems continuously fail to achieve targets for a good ecological status defined by the European Union due to eutrophication (Stutter et al., 2018). Of the major nutrients, carbon, nitrogen and phosphorus, phosphorus represents the limiting factor in most European streams. Therefore, the limitation and management of phosphorus is a key factor for the objective of mitigating eutrophication potential (Weigelhofer et al., 2018; Jarvie et al., 2013).

1.2 Phosphorus

Natural phosphorus mainly emanates from weathering rocks, terrestrial soils and decomposing organisms (Weigelhofer et al., 2018). In most situations, the amounts from these sources are minor; however, combined with anthropogenic sources, the overall amounts may become critical. Anthropogenic sources can be divided into point sources, which are geographically bound to one pollution point, and diffuse sources, which are geographically scattered. Point sources, especially wastewater treatment facilities, have a high potential for soluble reactive phosphorus (SRP) contamination; however, as a point source, they are relatively straightforward to control. Diffuse inputs from agricultural land use make the control of nutrient transport considerably more difficult (Withers and Jarvie, 2008). Here, intensive arable farming especially results in increased runoff and erosion of soils and nutrients from field to stream. These effects, combined with increased use of fertilizers, can lead to high phosphorus concentrations in agriculturally affected streams (Weigelhofer et al., 2018; Ockenden, 2012).

Phosphorus can occur in four different major forms: dissolved organic, particulate organic, dissolved inorganic and particulate inorganic phosphorus (Weigelhofer et al., 2018). There are multiple ways for phosphorus to react and transform among these forms. Phosphorus can be assimilated into biomass or mineralized from its biologically bound form back to its former soluble form. Most relevant to this master’s thesis, it can adsorb and desorb via geochemical and physical sorption processes with sediment particles (House, 2003).

1.3 Buffer Mechanisms

The physical adsorption and desorption processes mentioned above are mostly rapid and reversible. Dissolved phosphorus is adsorbed to the surfaces of aluminium hydroxides, iron hydroxides or alumino-silicates. The mechanisms are electrostatic extraction, ligand exchange or anion competition for the surface area of sediment particles (Withers and Jarvie, 2008). Soluble phosphorus is also exchanged via diffusion between concentration gradients of sediment phosphorus and water-column phosphorus. Thus, a higher concentration of phosphorus in the water column than in the sediment leads to adsorption to the sediment and vice versa to desorption into the water column. The result of this adsorption–desorption process is the potential ability of sediments to stabilize phosphorus input and output fluctuations within the stream (Froelich, 1988; Lottig and Stanley, 2007). The retention of phosphorus by abiotic sorption as well as by biotic assimilation is known as the phosphorus buffer mechanism (Froelich, 1988).

The phosphorus buffering capacity of streams can be affected by sediment particle size, which has been investigated extensively. The overall uptake capacity of sediments decreases with increasing particle size (Klotz, 1988; Munn and Meyer, 1990; McDaniel et al., 2009). As shown in Figure 1.1, Lottig and Stanly were able to extend this knowledge by adding the component of biotic or abiotic phosphorus uptake; more dominant abiotic uptake potential of stream sediments was observed with smaller particle sizes (Lottig and Stanley, 2007).

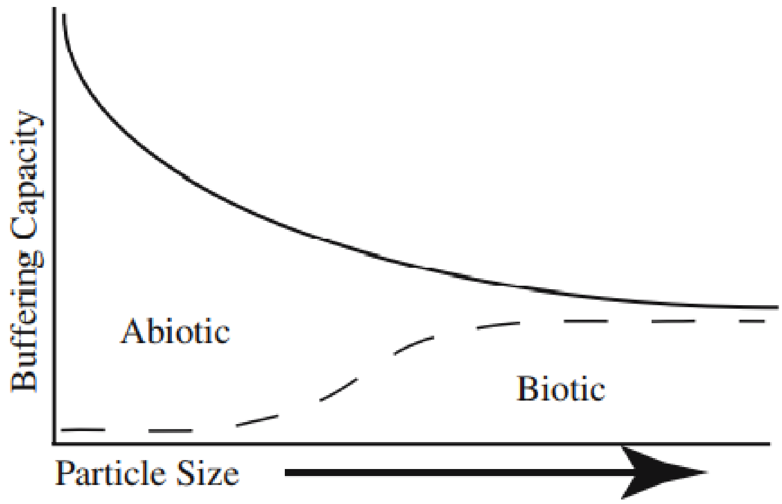

Figure 1.1: This diagram visualizes the uptake potential and mechanisms of benthic sediments across a particle size gradient. The solid line shows the overall uptake along the particle size gradient; the dashed line shows the transition between dominantly abiotic or biotic uptake (Lottig and Stanley, 2007).

Besides particle size, further conditions such as redox conditions and pH values can affect the abundance of surface area available for sorption processes. Channel hydrology and morphology also affect opportunities for phosphorus sorption due to their control of the residence time of water in the sediment (Withers and Jarvie, 2008).

1.4 Equilibrium Phosphorus Concentration

The sediment phosphorus concentration at which there is no net exchange of phosphorus between the water column and the sediment is called the equilibrium phosphorus concentration (EPC_0) (Froelich, 1988). Measurement of EPC_0 is a common tool to assess the phosphorus sorption characteristics of streams and rivers. A disequilibrium between phosphorus concentrations in the sediment and the water column can originate due to different reasons. For instance, the interaction between the sediment and the water may be weak, the sediment may have been disturbed recently, the sediment may be saturated with phosphorus or the phosphorus concentration in the water may have increased or decreased recently (Stutter and Lumsdon, 2008). If the disequilibrium exhibits higher concentrations in the water than in the sediment, the sediment acts as a phosphorus sink; if the water concentration is lower than the sediment concentration, the sediment acts as a phosphorus source (McDaniel et al., 2009). Simpson et al. showed in a meta-analysis that most stream sediments are not in equilibrium but rather constitute either a sink or a source of phosphorus (Simpson et al., 2021). In general, one can assume that sediments with low EPC_0 have more capacity for sorption than sediments with high EPC_0. Thus, a lower EPC_0 indicates a higher phosphorus buffering capacity (McDaniel et al., 2009).

1.5 Riparian Buffer Strips

Riparian buffers are a measure to mitigate diffuse nutrient inputs, especially from agricultural areas. They consist of vegetation longitudinal to the stream and aim to retain nutrients in the terrestrial system.

Riparian buffer strips can have many forms and appearances. They perform multiple key functions such as the trapping of nutrients, soil particles and pesticides that are transported from the field to the stream through ground and surface water. Furthermore, they contribute to biodiversity and water regulation (Stutter et al., 2012; Allan, 2004; Osborne and Kovacic, 1993). Regarding water regulation, other studies from Lower Austria have shown that riparian buffer strips lead to increased hydrological retention of nutrients in stream sediments (Weigelhofer et al., 2012). Those functions, especially the nutrient retention, is greater in riparian buffer strips containing bushes and trees than in those without them. This effect is due to larger habitats in the riparian zones and greater trapping of nutrients and soils in more intensely vegetated strips (Sweeney et al., 2004; Cao et al., 2018).

1.6 Land Use

In this thesis, land use has been divided into two types of agriculture: cropland and pasture. In Figure 1.2, the densities of both land use types are shown. Noticeable from these maps are the differences between north and south for both cropland and pasture. The high-density area, stretching from west to east in both maps, is positioned immediately below the Danube River, thus in the northern *Mostviertel*.

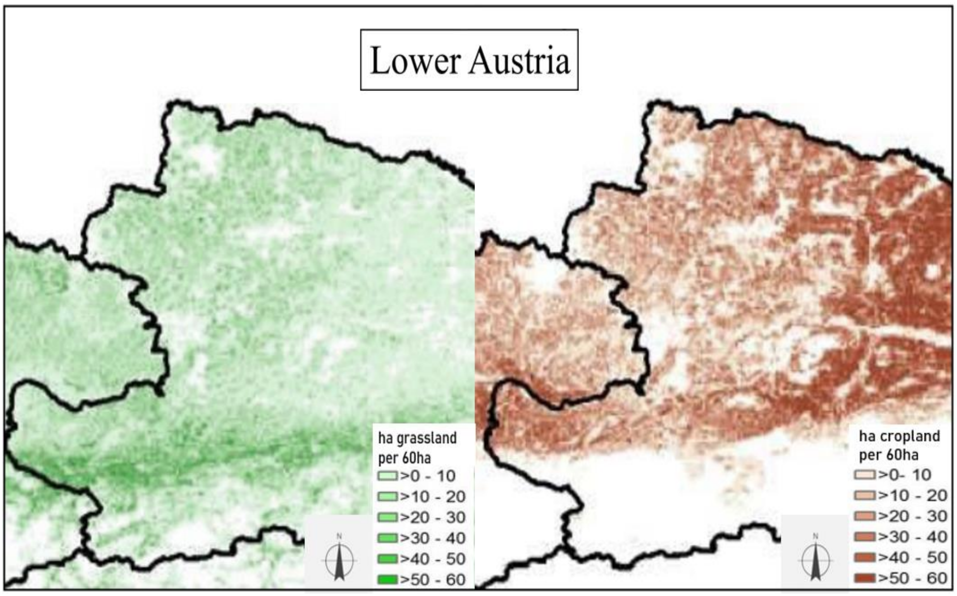

Figure 1.2: Two maps of the western parts of Lower Austria, showing the density of grasslands (left map) and the density of croplands (right map). Each element in the map includes 60 ha of land. The legends of each map show the density of each type of land use per element (AGES, 2018).

Cropland and pasture usually involve different practices for many agricultural processes; especially important for this thesis are fertilization practices. The Austrian ministry responsible for agriculture provides references for appropriate phosphorus levels in soils. Following this guideline, soils for pasture land use would be adequate with 47–68 mg P kg^{-1}; soils for crop land use would be adequate with 47–111 mg P kg^{-1}. These values demonstrate the differentiated phosphorus fertilization strategies based on land use. (Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, 2017).

Next to the nutrient loads, intensified agriculture has further negative effects on streams. Large amounts of soil transported from field to stream (Poeppl et al., 2019) and a tendency for smaller sediment particle sizes (Weigelhofer et al., 2018). To provide a perspective of soil erosion in an agricultural area, the Fugnitz catchment in the *Waldviertel* has soil loss rates from 0.28 to 11.7 t ha^{-1} y^{-1}. These amounts of erosion are due in part to differences in soil cultivation methods. In this regard, cropland can have ploughed, uncovered soil while pasture is mostly covered and thereby less exposed to erosion (Klik, 2004). Erosion has negative effects on agriculture, such as reduction of soil depth and a decrease in productivity. Negative effects on the stream are eutrophication, the clogging of spawning habitats and sedimentation of reservoirs (Poeppl et al., 2019). The dynamic of higher adsorption capacities at smaller sediment grain sizes (see Section 1.4) must be considered in the case of intensive agricultural land use. In this case, increased input of silt and clay fractions might lead to conglomeration and thereby to less sediment surface area and less residence time of the bypassing water. Furthermore, the newly introduced particles are likely to be loaded with phosphorus already; thus, this smaller grain size would not necessarily lead to larger buffering capacities (Nogaro et al., 2010; Jarvie et al., 2005). With more soil and nutrients eroding from intensive agricultural land use in cropland, one might expect riparian buffers to have an even higher effect on nutrient retention in these areas than they would have in pasture-associated land use.

1.7 Flood Conditions

Floods can have drastic effects on the afore described phosphorus dynamics between sediment and water column. Surface runoff due to flooding of the stream bank and the riparian zone increases erosion and transport of phosphorus into the stream (Macrae et al., 2011). Jarvie et al. (2005) have shown that benthic sediments have the highest phosphorus buffering capacity in conditions of low flow. A higher share of sediment surface per volume of flowing water and an extended time of contact between sediment and water are the causes of these effects (Jarvie et al., 2005). Considering the sampling design of 1 km without followed by 1 km with riparian vegetation (see chapter 2.1), one might not expect a difference between the two stretches, as suspended material would be distributed throughout the entire area.

1.8 Research Objectives

In this thesis, phosphorus retention by benthic sediments and the effect of riparian buffer strips on phosphorous buffering capacity were investigated. Specifically, it was intended to determine whether the sediment phosphorus retention of a stream is improving from a 1-km stretch without riparian vegetation to a consecutive 1-km stretch with riparian vegetation. All streams were sampled in order to identify the differences between areas with riparian vegetation and areas with no riparian vegetation. To be able to obtain a realistic impression of agricultural streams in the federal state of Lower Austria, the two major land use types, cropland and pasture, were included. The intention was to determine whether different agricultural land use types exhibit differences in phosphorus retention between areas with riparian vegetation and areas with no riparian vegetation. Furthermore, an additional comparison to flood scenarios became of interest and parameters such as respiration and sediment grain sizes were evaluated to gather insights into bacterial activity and sediment qualities.

Multiple questions were explored in this thesis: (i) Do riparian buffer strips increase the phosphorus retention capacities of stream sediments compared to sites without riparian buffer strips? (ii) Do riparian buffer strips increase or decrease sediment respiration compared to sites without riparian buffer strips? (iii) Do riparian buffer strips affect sediment grain sizes, organic matter and nutrient contents of the sediments compared to sites without riparian buffers? (iv) Does cropland increase the previously questioned effects (i, ii, iii) compared to pasture? (v) Do the previously questioned effects (i, ii, iii) differ between the two regions of *Waldviertel* and *Mostviertel*? (vi) How do flood conditions alter the effects which were previously questioned (i, ii, iii)?

Hypotheses:

1) Phosphorus retention capacities of stream sediments are greater in sites with riparian buffer strips than in sites without riparian buffer strips.
2) Sediment respiration is greater in sites without riparian buffer strips than in sites with riparian buffer strips.
3) Sites with riparian buffer strips have larger sediment grain sizes and lower organic matter and sediment nutrient contents than sites without riparian buffer strips.
4) The effects specified in Hypotheses 1, 2 and 3 are greater in crop- than in pasture-dominated land use.
5) The effects specified in Hypotheses 1, 2 and 3 are indifferent to flood conditions.

2. Methods

This thesis has investigated the sediment buffering mechanism of adsorption and desorption and the effects that riparian vegetation can have on this process. Biological assimilation through algal and microbial biofilms could not be excluded by the methods which were applied.

2.1 Site Descriptions

The main objective of this thesis was to determine the differences between the effects of riparian vegetation and lack of riparian vegetation on sediment buffer capacities. To investigate this difference, eight streams were chosen; these streams included a section of at least 1 km predominantly without riparian vegetation directly followed by a section of at least 1 km with riparian vegetation present. All streams are located within the federal state of Lower Austria (Austria), with four in the Mostviertel and four in the Waldviertel. The names of the streams are *Thaya, Zwettl, Purzelkamp, Große Taffa, Prollingbach, Brettlbach, Schlattenbach* and *Dollbach*. All streams have been further characterized in Table 1. In addition to the immediate vegetation, the influence of the dominant surrounding land use was investigated. All streams have predominantly agricultural surroundings. Therefore, surrounding land use was categorized as cropland-dominated agriculture and pasture-dominated agriculture. The *Mostviertel* and the *Waldviertel* each have two pasture- and two crop-field-surrounded streams.

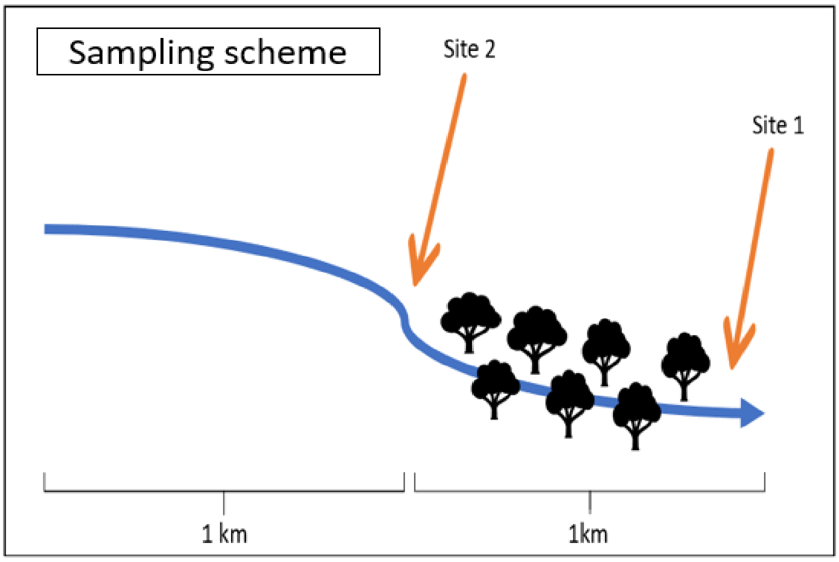

Figure 2.1: Sampling scheme valid for all considered streams. Each stream has two sections: one without riparian vegetation (1 km) followed downstream by another (1 km) with riparian vegetation. Site 1 was positioned at the end of the considered stream section; site 2 is positioned between the two sections.

Table 2.1: Characterization of stream sites and their physical conditions. Riparian-vegetated sites were designated as site 1 and sites without riparian vegetation as site 2.

Stream	Coordinates Site 1	Coordinates Site 2	Region	Land use	Discharge (m^3/s)	Width Site 1 (m)	Width Site 2 (m)
Thaya	48°44'32.6"N 15°12'16.4"E	48°44'50.1"N 15°11'47.0"E	*Waldviertel*	Pasture	1.88	3.1	3.5
Zwettl	48°37'09.5"N 15°03'42.0"E	48°37'23.6"N 15°03'19.0"E	*Waldviertel*	Pasture	3	6.2	6.1
Purzelkamp	48°31'15.7"N 15°14'36.1"E	48°30'52.5"N 15°14'22.8"E	*Waldviertel*	Crop	1	3.9	3.1
Große Taffa	48°40'41.9"N 15°37'39.0"E	48°40'56.7"N 15°36'51.5"E	*Waldviertel*	Crop	0.2	2.78	2.37
Prollingbach	47°53'52.5"N 14°52'57.9"E	47°53'26.3"N 14°52'37.3"E	*Mostviertel*	Pasture	0.18	2.5	1.7
Brettlbach	47°57'21.4"N 15°02'27.6"E	47°57'16.9"N 15°03'11.9"E	*Mostviertel*	Pasture	0.14	1.9	1.33
Schlattenbach	48°07'54.9"N 15°15'40.0"E,	48°07'56.9"N 15°14'55.0"E	*Mostviertel*	Crop	0.04	2.13	1.1
Dollbach	48°09'53.6"N 15°11'01.1"E	48°09'25.9"N 15°11'05.0"E	*Mostviertel*	Crop	0.03	1.05	1.21

To provide a geographical image of the stream sites, Figure 2.2 shows all sites on a map of Lower Austria.

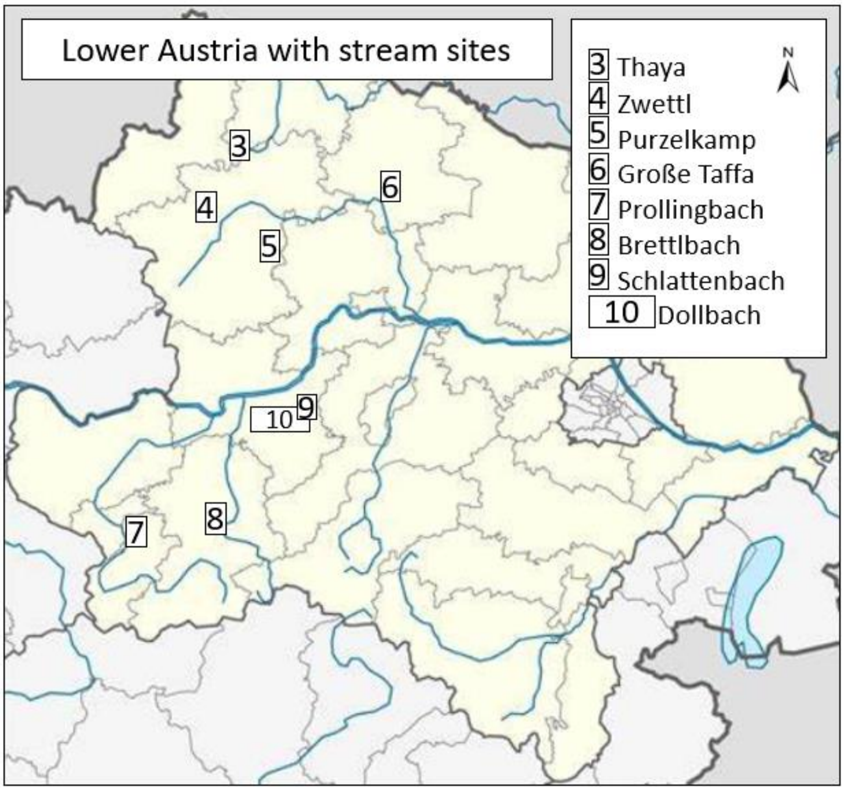

Figure 2.2: Map of Lower Austria with stream sites. Streams 3, 4, 5 and 6 are positioned in the Wadlviertel and streams 7, 8, 9 and 10 in the Mosteviertel. Streams 3, 4, 7 and 8 are surrounded by pasture and streams 5, 6, 9 and 10 by crop-dominated land use.

The research focus of this thesis was restricted to two regions within Lower Austria, the *Mostviertel* and the *Waldviertel*. In Figure 2.3, the differences between these regions have been emphasized. The left map has been provided as an orientational aid for the comparison of the two regions. Combined with the map on the right, the geological settings and topographical properties can be visualized.

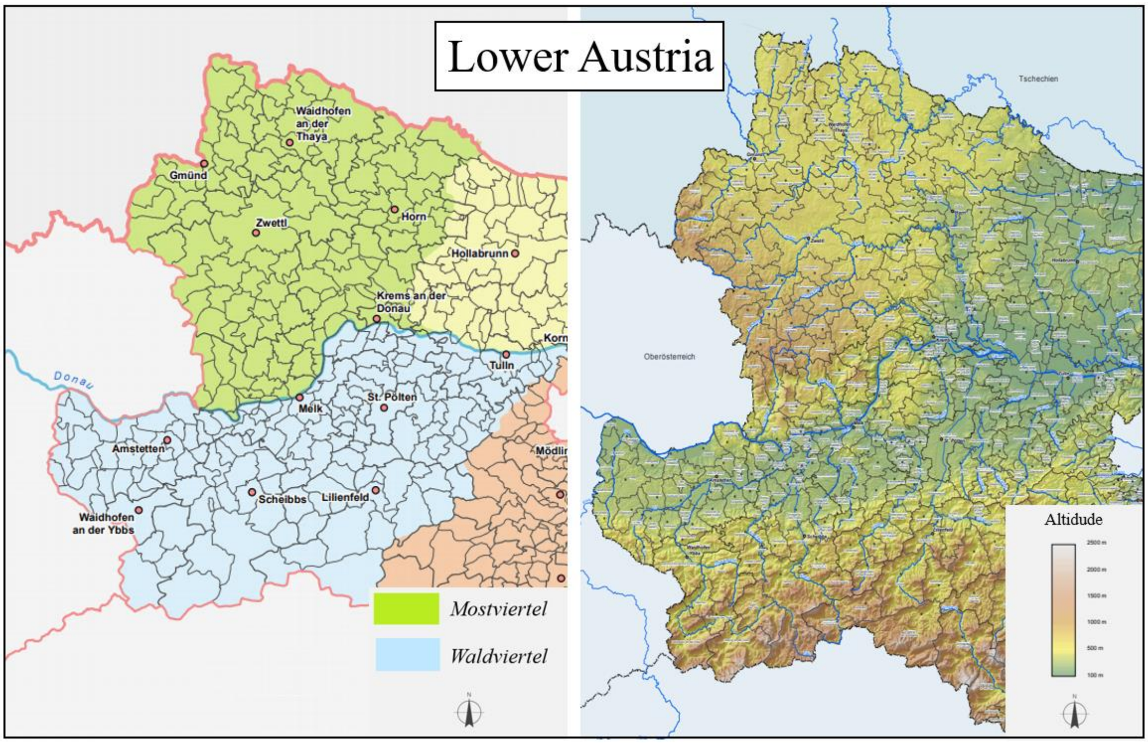

Figure 2.3: The two maps both depict Lower Austria. The left map shows the domain of the two regions *Waldviertel* and *Mostviertel.* The map on the right shows the difference in altitude between the two regions (Amt der Niederösterreichischen Landesregierung, 2017).

The *Mostviertel* is geologically quite diverse. In the north, it starts with sedimentary depositions from the Danube River, transitioning to zones of molasse to the south. Further into the foothills of the Alps, the Flyschzone begins and continues with the Calcareous Alps. The *Waldviertel* geology begins to the south with the same sedimentary dispositions of the Danube River; further north it is mainly the Bohemian Massif. This massif is a mixture of granite, granulite, orthogneiss, paragneiss and migmatite (Geologische Bundesanstalt). Above this foundation, the *Waldviertel* contains mainly brown earth and pseudogley, with some podzol to the west. The *Mostviertel* has brown earth and pseudogley in the north, transitioning to rendzina with increasing altitude to the south (NÖ Agrarbezirksbehörde, 2016).

2.2 Sampling Methods

In this section, the methods for obtaining the samples are described. Sediment and water samples were taken from sites 1 and 2 of each chosen stream.

2.2.1 Sediment Samples

Sediment samples used for the determination of the EPC_0 of the sediments, sediment respiration, sediment phosphorus concentration, organic matter and grain size distribution were collected on four sampling dates. The *Thaya*, *Zwettl* and *Purzlkamp* were sampled on 24 June 2020. The *Große Taffa* and the *Prollingbach* were sampled on 1 July 2020 and the *Brettlbach*, *Dollbach* and *Schlattenbach* on 10 July 2020. All streams were in a condition of low flow on the sampling date. To provide a contrast to flood conditions, the *Schlattenbach* and the *Dollbach* were additionally sampled on 15 June 2020 to further distinguish between flood and no flood. The sampling procedure ensured constant conditions throughout all streams and sites. Sediments were collected in the centre of the stream; only the upper 4 cm were sampled, as this is the domain of phosphorus exchange between the water column and sediment (Stutter and Lumsdon, 2008). At each site, five replicates were taken and sieved through a 4 mm filter. Five additional, unsieved replicates were taken for analysis of grain size distribution. All samples were placed in plastic bags, cooled to approximately 4 °C for transportation and stored at 4 °C.

2.2.2 Water Samples

To analyze the EPC_0 results from the perspective of being a phosphorus sink or source, water phosphorus concentrations were obtained. Further nutrients, carbon and nitrogen, were additionally analyzed as descriptive parameters. Water samples were collected at all sites on 26 August 2020. All samples were filtered on-site with muffled GF/F filters (0.7 µm). The filtered water samples were then cooled to approximately 4 °C for transportation. In the laboratory, all samples were stored at 4 °C.

2.3 Analytical Methods

2.3.1 Equilibrium Phosphorus Concentration

The equilibrium phosphorus concentration (EPC_0) determines the phosphorus adsorption characteristics of sediments and has been the subject of many studies (e.g. FROELICH, 1988; Stutter and Lumsdon, 2008; Weigelhofer et al., 2018; McDowell, Sharpley and Folmar, 2003). For the experiments described here, a six-batch equilibrium experiment was chosen using concentrations of 0, 15, 30, 60, 120 and 250 µg P L^{-1}. From each of the five sieved replicas per site, six 10-g portions were extracted with 50 ml of solution at the six different concentrations. The substances used to attain the different concentrations were monosodium phosphate (NaH_2PO_4) and untreated groundwater. Untreated groundwater was used due to its similar ionic composition to the study streams (Weigelhofer et al., 2018). The prepared samples were positioned on a gentle shaker in a dark climate chamber at a constant temperature of 20 °C. After 24 hours the samples were centrifuged for 15 minutes at 3000 rpm. The supernatant was then assessed for SRP concentrations by continuous-flow analysis (CFA, Systema Analytical Technology). The calculation for the quantity of phosphorus being adsorbed or desorbed per gram of sediment was calculated following Weigelhofer et al. (2018):

$$P_{sorp} = \frac{(\boldsymbol{SRP_{ini}} - \boldsymbol{SRP_{fin}}) * \boldsymbol{V}}{\boldsymbol{w_{sed}}}$$

P_{sorp}: Amount of desorbed or adsorbed SRP (µg L^{-1});
SRP_{ini}: Initial SRP concentration of the water (µg L^{-1});
SRP_{fin}: Final SRP concentration of the water (µg L^{-1});
V: Volume (L);
W_{sed}: Dry weight of the sediment (g).

The values of P_{sorp} were plotted against the associated SRP_{fin} data (Jarvie et al., 2005). The resulting data points were then fitted with a linear isotherm to calculate the intercept of the linear isotherm with the *x*-axis. This graph allows calculation of the EPC_0 values; an example calculation is depicted in Figure 2.3.

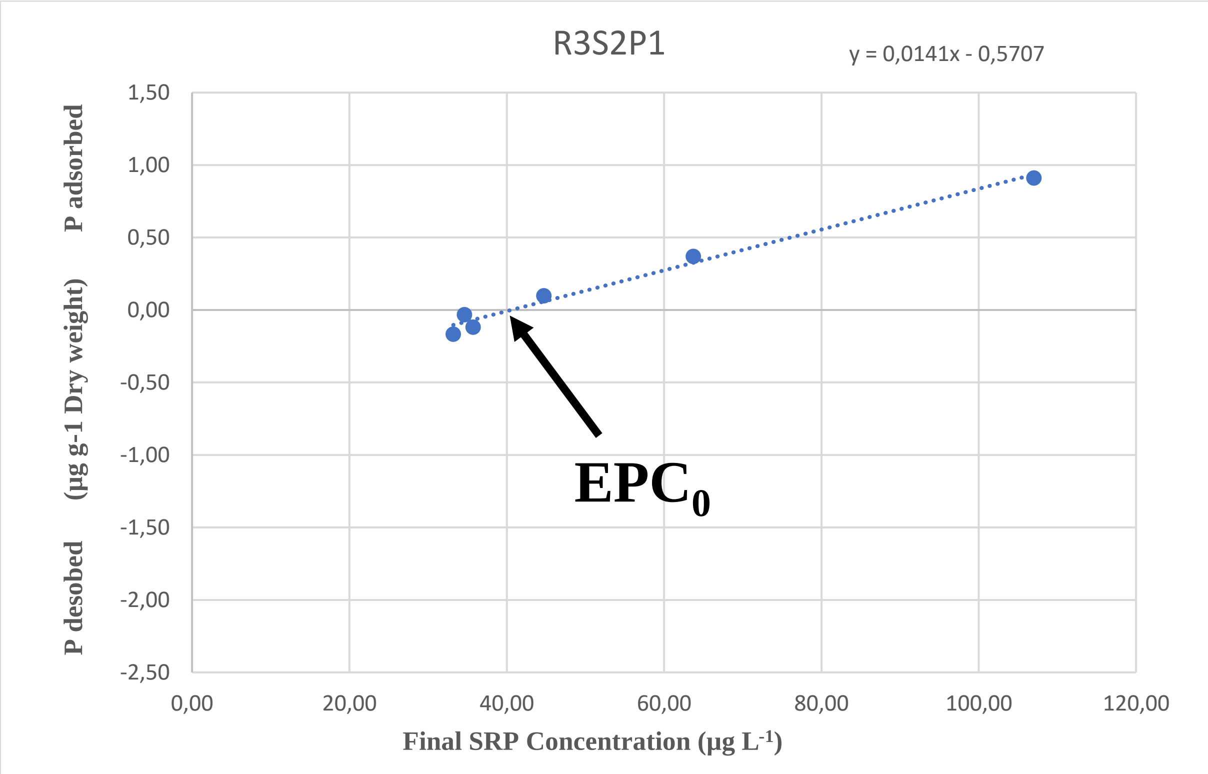

Figure 2.3: P_{sorp} plotted against SRP_{fin} within the same stream and site. The resulting trendline intercepts the zero point between desorption and adsorption, the EPC_0. Data from replica #3 of the Thaya site without riparian vegetation is presented. The *x*-intercept is 40.48 µg P L^{-1}.

The EPC_0 data were derived from experimentation with the sediment. To determine whether a site was a phosphorus source or a phosphorus sink, it was compared to the water phosphorus concentration at the same sites. If the stream water phosphorus was higher in concentration than the sediment EPC_0, the sediment was likely to act as a sink. If the sediment EPC_0 was higher than the stream water phosphorus concentration, the sediment was likely to act as a source (McDowell, Sharpley and Folmar, 2003). Therefore, the median of all EPC_0 values per site was compared to the water SRP.

2.3.2 Sediment Respiration

Respiration rates of the sediment samples were measured with optical oxygen sensor spots (Warkentin et al., 2007). As respiration in sediments occurs continuously, it permits no delay in analysis; therefore, these experiments were started on the same day as the sampling. As mentioned previously, five 4-mm-sieved replicas were taken at each site. From each replica, 10–20 g of sediment were incubated with untreated groundwater. The oxygen concentration was then measured immediately and in the following 24 hours, with intervals of four hours (0, 4, 8, 12, 16, 20, 24 hours). During the experiment, all samples were positioned on a gentle shaker in a dark climate chamber with a constant temperature of 20 °C. The data obtained, in mg oxygen per litre, were then plotted against time and represented by a regression line. The slope of the regression line resulted in a suitable value for the comparison of declining oxygen concentrations between the streams and sites.

2.3.3 Sediment Phosphorus Concentration

Inorganic Phosphorus

The extraction of phosphorus with hydrogen chloride (HCl) incorporates carbonate-bound phosphorus (Ruban et al., 2001). From each site, three replicas of sieved (4 mm) sediment were mixed with 1 M HCl. The samples were then shaken for 16 hours in the dark at a constant temperature of 20°C. Subsequently, all samples were centrifuged for 15 minutes at 3000 rpm. The supernatant was then assessed for SRP concentrations by continuous-flow analysis (CFA, Systema Analytical Technology).

Total Phosphorus

The following method was used to determine the amount of total phosphorus in the sediment samples, adsorbed abiotically to the sediment or assimilated biotically into biomass (Malá and Lagová, 2014). Three replicas of sieved (4 mm) sediments from each site were dried, ground and muffled in an oven at 450 °C for 4 hours. Then, 0.1–0.2 g of each replica was extracted and mixed with 0.5 M sulfuric acid (H_2SO_4). After 20 minutes of reaction time, the samples were placed into a microwave with 1600 W (CEM MarsXpress). The samples were then diluted with pure distilled water and assessed for SRP concentrations with continuous-flow analysis (CFA, Systema Analytical Technology).

2.3.4 Organic Matter Content

To measure dry weight, all five replicas were dried in an oven for 16 hours at 75 °C. The ash-free dry weight was then determined by combusting all replicas at 400 °C for 4 hours. The difference between dry weight and ash-free dry weight approximated the organic matter content.

2.3.5 Grain Size Distribution

Grain size distribution was determined via dry sieving following ÖNORM EN ISO 14688-1 (ÖSTERREICHISCHES NORMUNGSINSTITUT, 2013). The six different sieve sizes were 20 mm, 6.3 mm, 2 mm, 1 mm, 0.5 mm and 0.125 mm. Among all fractions, the sand fraction (<2 mm) and the silt and clay fraction (<0.063 mm) were of primary interest to this thesis. To estimate the silt and clay fraction, a cumulative proportion was deducted from the relative proportions of the fractions, starting with the finest. The sand fraction could thereby be calculated. To estimate the silt and clay fraction, the cumulative proportions were plotted against the sieve size. A linear isotherm between the point of origin and the 0.125 mm point allowed an estimation of the silt and clay grain size intercept (x = 0.063 mm).

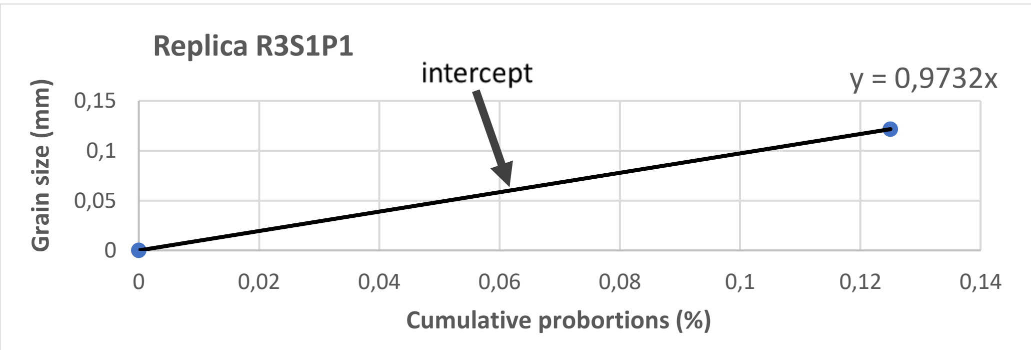

Figure 2.4: Example plot of the grain size against the cumulative proportions. Included is the intercept with 0.063 mm to estimate the silt and clay fraction (Replica R3S1P1 = 0.061 % silt and clay in the sediment sample).

2.3.6 Water Nutrients

The extracted water samples were analyzed for phosphorus in orthophosphate ($P\text{-}PO_4$), DOC, nitrogen in nitrogen dioxide ($N\text{-}NO_2$) and nitrogen in nitrate ($N\text{-}NO_3$) by continuous-flow analysis (CFA, Systema Analytical Technology).

2.4 Statistical Analysis

All data were processed with IBM SPSS statistics software. The normality of the data was tested using Kolmogorov–Smirnov and Shapiro–Wilk tests. The EPC_0 data were revealed not to be normally distributed. This condition restricted the statistical analysis to nonparametric tests. For data comparisons of EPO_0, respiration, grain size, organic matter and phosphorus in sediments, regarding both immediate vegetation (in site 1 and site 2) and surrounding dominant land use (cropland and pasture), Wilcoxon signed-rank tests were applied. For the investigation of each stream alone, Man–Whitney U tests were performed. To compare the differences between land use of all data for the parameters mentioned above, Kruskal–Wallis tests were performed.

3. Results

3.1 Equilibrium Phosphorus Concentration

For the comparison of all sites with riparian vegetation (S1/Veg) and without riparian vegetation (S2/NVeg), the medians of the five replicates per site were calculated. The median EPC_0 values, depicted in Figure 3.1, ranged from 15.3 µg L^{-1} to 83.7 µgL^{-1} for sites with riparian vegetation and from 23.2 µgL^{-1} to 158.7 µgL^{-1} for sites without riparian vegetation. Their averages were 40.3 µgL^{-1} for riparian-vegetated sites and 38.6 µgL^{-1} for non-riparian-vegetated sites. They did not differ significantly but tended toward higher concentrations at the riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.674, n = 16).

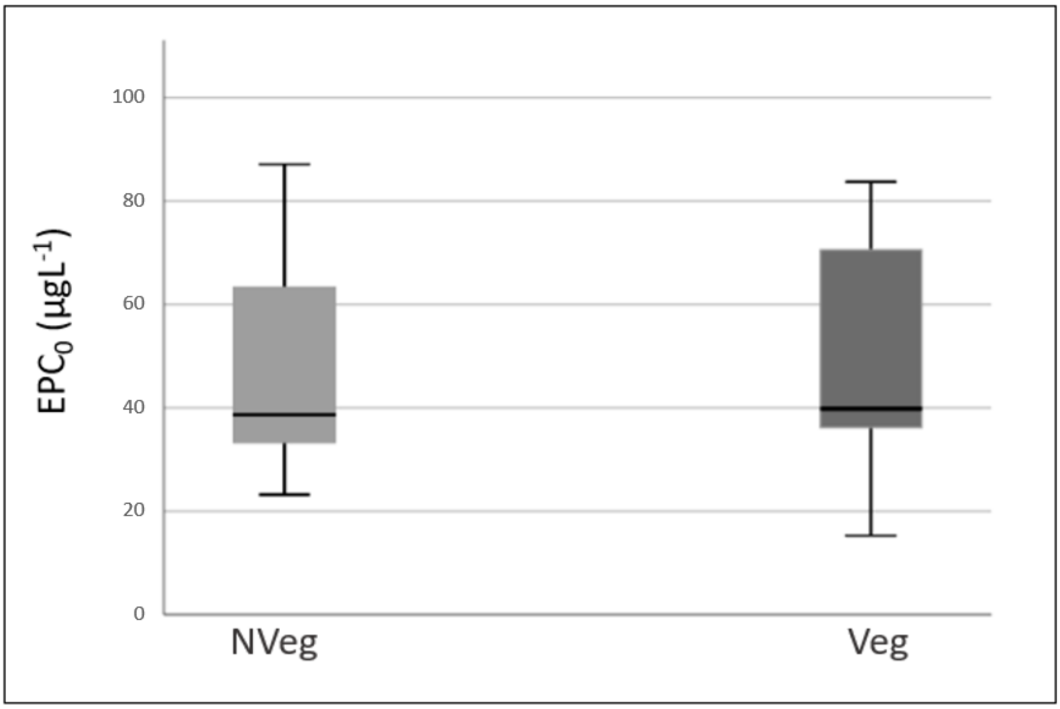

Figure 3.1: Median EPC_0 for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (n = 16).

A further analysis was undertaken by separating the data according to land use. Land used as pasture showed no significant difference between sites with and without riparian vegetation, with slightly higher concentrations at the riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.27, n = 8). Sites in areas with crop-dominated land use showed the contrary result. Here, EPC_0 medians of non-riparian-vegetated sites were slightly, although not significantly, higher (Wilcoxon signed-rank test, p = 0.06, n = 8).

Additional analysis of the differences between the land use types was performed using Kruskal–Wallis tests. A significant difference between the two agricultural land use types was found, with mean ranks of 51.13 for crop-dominated areas and 29.88 for pasture-dominated areas (Kruskal–Wallis test, X^2 = 16.725, p = 0.000, n = 80), suggesting a higher EPC_0 for all data from crop sites.

Paired comparisons of sites with and without riparian vegetation for each separate stream revealed significant differences for 4 out of 8 streams (Table 3.1). The pasture streams 3 and 7 exhibited significant differences in which riparian-vegetated sites showed higher EPC_0 concentrations than non-riparian-vegetated sites. Pasture stream 4 pointed in the same direction but without significance. Stream 8 showed significance but in the opposite direction, such that concentrations at non-riparian-vegetated sites were greater. It was found that all cropland streams, streams 5, 6, 9 and 10, showed higher concentrations at riparian-vegetated sites than at non-riparian-vegetated sites. However, only the result of stream 9 showed significance toward this tendency (Mann–Whitney U test, n = 10).

Table 3.1: Comparison of EPC_0 data between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U Test	Exact Sig. [2*(1-tailed Sig.)]	Mean rank S1	Mean rank S2
Stream 3 (Pa)	0.008	8	3
Stream 4 (Pa)	0.841	5.8	5.2
Stream 5 (Cr)	0.222	4.2	6.8
Stream 6 (Cr)	0.841	5.2	5.8
Stream 7 (Pa)	0.008	8	3
Stream 8 (Pa)	0.008	3	8
Stream 9 (Cr)	0.008	3	8
Stream 10 (Cr)	0.222	4.2	6.8

Figure 3.2 shows the relationship between the median of EPC_0 and water SRP concentrations of each site. This relationship can reveal sites to be phosphorus sinks or sources or in equilibrium between the two concentrations. The diagonal in the plot delimits the sites between sinks and sources of phosphorus. Two additional dashed lines of +/– 20% from the diagonal have been added on each side to allow for a certain variability of the data. All sites within those lines were considered to be in equilibrium between the EPC_0 and the water SRP concentrations. Sites above or below the dashed lines were considered to be phosphorus sources or sinks, respectively (Weigelhofer et al., 2018; Jarvie et al., 2005). This data showed no site to be a phosphorus sink. The three sites within equilibrium were both sites of stream 5 and the riparian-vegetated site of stream 4. All other sites were sources, with the non-riparian-vegetated site of stream 9 as the outlying highest source.

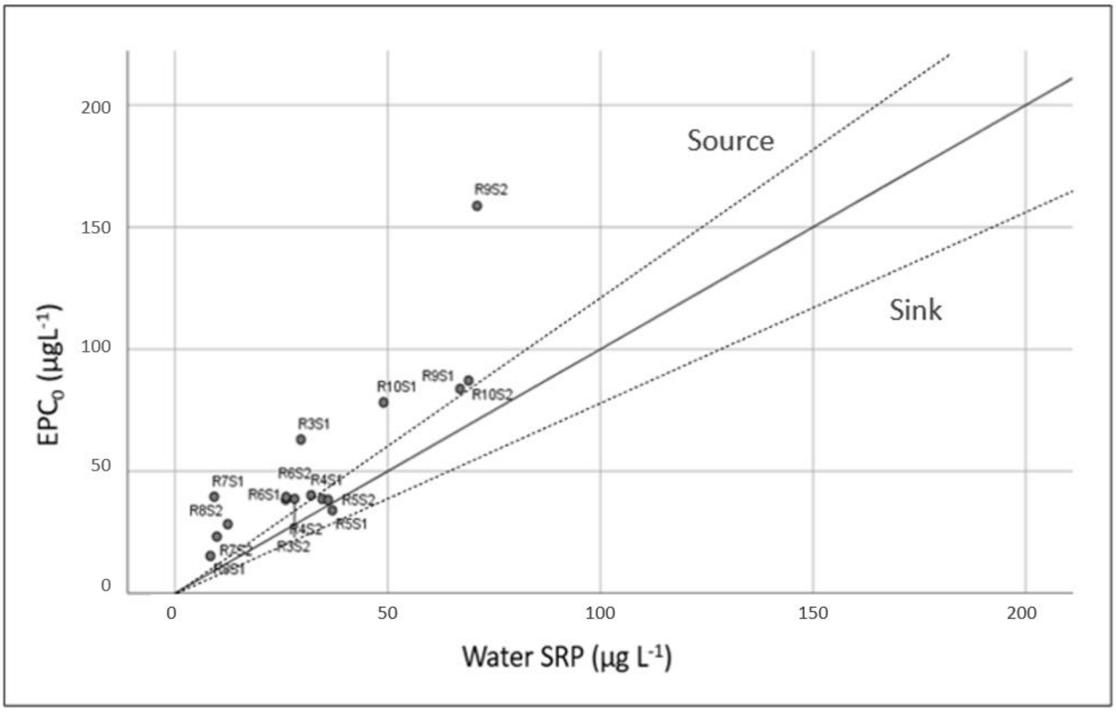

Figure 3.2: Relationship between the median of EPC_0 and water SRP concentrations of each site. The diagonal shows the probability of a site as a sink or a source of phosphorus. The dashed lines were placed at +/– 20 % of the diagonal. They show the area in which sites are in equilibrium between the two concentrations. Each dot resembles a stream site where 'R' and the following number stand for the stream number and the subsequent 'S1' or 'S2' stands for a riparian-vegetated or non-riparian-vegetated site of the stream.

The differences between the two concentrations were further analyzed with the two boxplots depicted in Figure 3.3. It is visible that riparian-vegetated sites were more likely to act as a source than non-riparian-vegetated sites. However, this disparity was statistically insignificant (Wilcoxon signed-rank test, p = 0.779, n = 16).

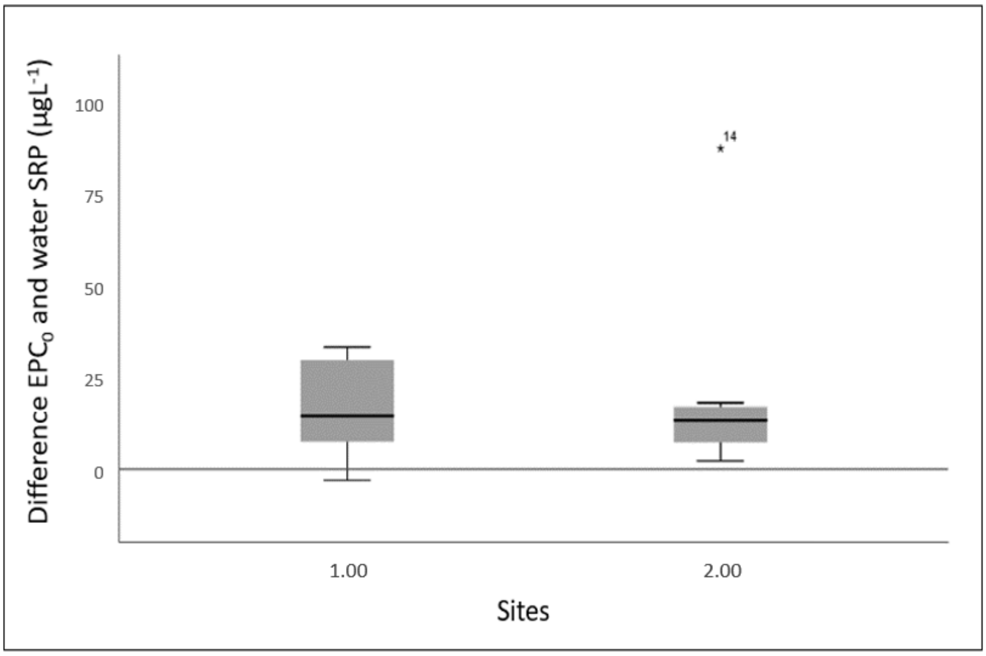

Figure 3.3: Boxplots of the probability of a site to act as a source or a sink of phosphorus, classified by riparian-vegetated (1.00) and non-riparian-vegetated sites (2.00). Positive or negative values indicate that sediments act as a source or a sink, respectively. Shown are the maximal and minimal values, the 25 % and 75 % quantiles as well as the medians (n = 16).

The data were further analyzed to examine differences between the regions of *Waldviertel* and *Mostviertel*. Both regions showed no significant differences but higher EPC_0 at riparian-vegetated sites (*Waldviertel*: Wilcoxon signed-rank test, p = 0.365, n = 8; *Mostviertel*: Wilcoxon signed-rank test, p = 0.465, n = 8). A more general investigation performed without separating riparian vegetation showed all sites of the *Mostviertel* to have higher EPC_0 values than the sites of the *Waldviertel*, without significance (Kruskal–Wallis test, X^2 = 1.060, p = 0.303, n = 80).

The *Schlattenbach* and the *Dollbach*, which are both located in the *Mostviertel* region, were sampled during flood and low-flow conditions. Statistical analysis showed that the streams during flood conditions, with both sites included, tended toward higher EPC_0 than during low-flow conditions (Wilcoxon signed-rank test, $p = 0.715$, $n = 8$). Examining all of the data from flood and low-flow conditions separately, both tended to have a higher EPC_0 at non-riparian-vegetated sites, but without significant difference (flood: Wilcoxon signed-rank test, $p = 0.180$, $n = 4$; low flow: Wilcoxon signed-rank test, $p = 0.180$, $n = 4$). By comparing flood and low-flow conditions within all sites with riparian vegetation, it became visible that the data tended toward higher EPC_0 during conditions of low flow (Wilcoxon signed-rank test, $p = 0.180$, $n = 4$). The same comparison between flood and low flow within non-riparian-vegetated sites showed that EPC_0 tended to be higher during flood conditions (Wilcoxon signed-rank test, $p = 0.655$, $n = 4$).

3.2 Respiration

To compare the oxygen consumption of sites with riparian vegetation (Veg) and sites without riparian vegetation (NVeg), the medians of all five replicas of each site were taken. Depicted in Figure 3.4, the boxplots showed values between 0.17 mg g^{-1} h^{-1} to 0.81 mg g^{-1} h^{-1} for the riparian-vegetated sites and 0.16 mg g^{-1} h^{-1} to 0.74 mg g^{-1} h^{-1} for the non-riparian-vegetated sites. The average of those medians was 0.25 mg g^{-1} h^{-1} (Veg) and 0.23 mg g^{-1} h^{-1} (NVeg). The difference showed no statistical significance; however, the results tended toward higher respiration rates at the riparian-vegetated sites (Wilcoxon signed-rank test, $p = 0.313$, $n = 16$). This effect translates to lower respiration at sites with no riparian vegetation and higher respiration at sites with riparian vegetation.

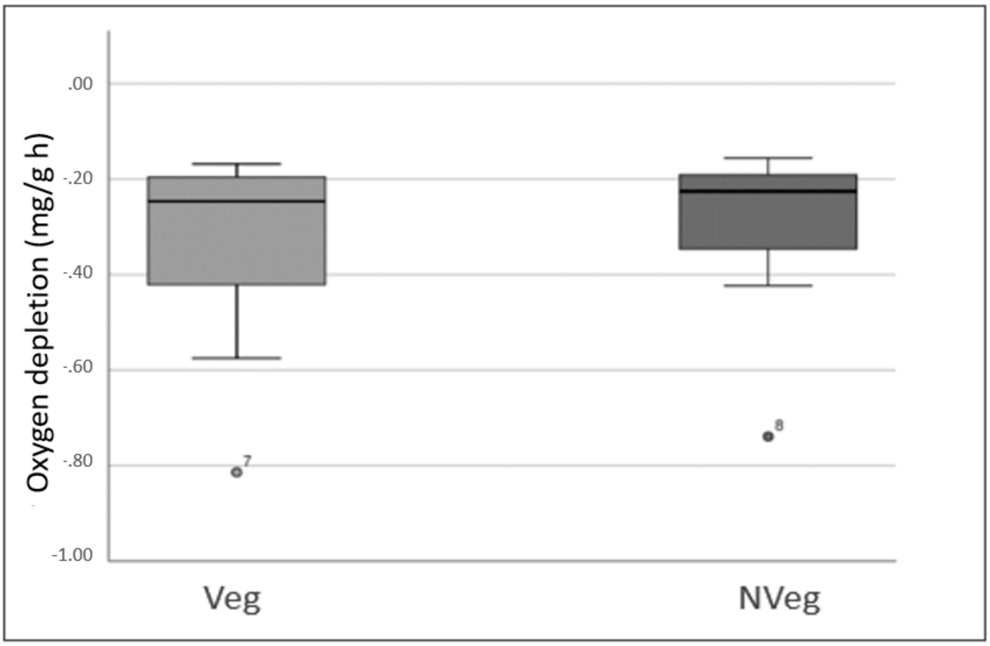

Figure 3.4: Medians of oxygen consumption over time for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (*n = 16*).

To determine differences in respiration according to land use, the data were further analyzed. All data for pasture land use showed no significant difference but slightly higher oxygen consumption at the riparian-vegetated sites compared to the sites without riparian vegetation (Wilcoxon signed-rank test, p = 0.250, n = 8). All data from crop land use showed similar results. No significance was observed, but the results tended toward higher oxygen consumption at sites with riparian vegetation (Wilcoxon signed-rank test, p = 0.875, n = 8).

A Kruskal–Wallis test was applied to determine differences between the land use types, without separation from riparian vegetation. It showed no significant difference, with cropland streams producing slightly higher respiration results than the pasture streams (Kruskal–Wallis test, X^2 = 0.025, p = 0.875, n = 80).

To compare each stream alone, paired comparisons were applied via Mann–Whitney U tests, shown in Table 3.2. As the data were included with negative values, higher mean ranks indicate lower oxygen consumption and vice versa. Only stream 4 showed significant differences, with riparian vegetation having higher ranks. The same tendency but without significance was also exhibited by streams 3, 6 and 7. The other streams (5, 8, 9 and 10) tended to have higher respiration values at non-riparian-vegetated sites (Mann–Whitney U test, n = 10).

Table 3.2: Comparison of respiration data between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U Test		Exact Sig. [2*(1-tailed Sig.)]	Mean rank S1	Mean rank S2
Stream 3	(Pa)	0.151	4	7
Stream 4	(Pa)	0.008	3	8
Stream 5	(Cr)	0.690	6	5
Stream 6	(Cr)	0.310	4.4	6.6
Stream 7	(Pa)	0.056	3.6	7.4
Stream 8	(Pa)	0.421	6.4	4.6
Stream 9	(Cr)	0.151	7	4
Stream 10	(Cr)	1.000	5.6	5.4

Differentiation of the data by region and riparian vegetation showed, without significance, higher respiration at the riparian-vegetated sites of both regions (*Waldviertel*: Wilcoxon signed-rank test, p = 0.447, n = 8; *Mostviertel*: Wilcoxon signed-rank test, p = 0.447, n = 8). An investigation of which region had overall higher values revealed the *Waldviertel* sites had higher respiration values than the *Mostviertel* sites, again without significance (Kruskal–Wallis test, X^2 = 3.503, p = 0.061, n = 80).

The comparison of flood samples with all low-flow samples resulted in coequal results, in which neither of the conditions showed a statistical difference (Wilcoxon signed-rank test, p = 1.000, n = 8). Within the flood conditions, riparian vegetation tended toward higher respiration (flood: Wilcoxon signed-rank test, p = 0.180, n = 4). Similarly, low-flow conditions also tended toward higher respiration results in riparian-vegetated sites (low flow: Wilcoxon signed-rank test, p = 0.180, n = *4*). If flood and low-flow conditions were compared within all sites with or without riparian vegetation, a diverse result emerged. In sites with riparian vegetation, the low-flow conditions tended toward higher respiration rates while in sites without riparian vegetation the flood conditions tended toward higher respiration (riparian vegetation: Wilcoxon signed-rank test, p = 0.180, n = 4; no riparian vegetation: Wilcoxon signed-rank test, p = 0.180, n = 4).

3.3 Organic Matter

A comparison concerning organic matter content was made between sites with riparian vegetation and sites without riparian vegetation. The medians of all five replicas are depicted in Figure 3.5. The values of the medians ranged from 0.02 g to 0.26 g for riparian-vegetated sites and from 0.03 g to 0.09 g for non-riparian-vegetated sites. The average of the medians was 0.06 g for the riparian-vegetated sites and 0.03 g for the non-riparian-vegetated sites. These results, however, were not significantly different (Wilcoxon signed-rank test, p = 0.453, n = 16).

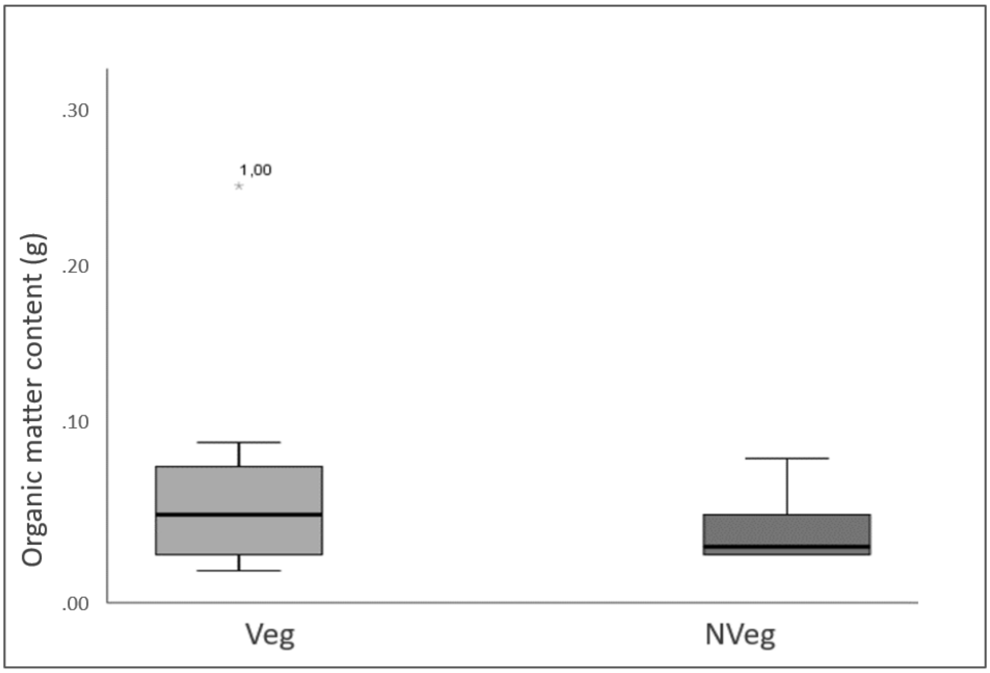

Figure 3.4: Medians of organic matter content for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (n = *16*).

Further differentiation of the data by land use produced varied results. The results from pasture-dominated land showed more organic matter at riparian-vegetated sites but without significance (Wilcoxon signed-rank test, p = 0.250, n = 8). In contrast, crop-dominated land use showed higher organic matter content at the non-riparian-vegetated sites, also without significance (Wilcoxon signed-rank test, p = 0.750, n = 8).

To separate the data without differentiating between the presence or absence of riparian vegetation, a Kruskal–Wallis test was applied. There was no significant difference in organic matter content detectable between cropland and pastures, while the results showed slightly higher content at crop-dominated land use sites (Kruskal–Wallis test, $X^2 = 3.344$, $p = 0.067$, $n = 80$).

Mann–Whitney U tests were applied to differentiate sites with riparian vegetation and sites without it, considering each stream separately. As shown in Table 3.3, only streams 3, 5 and 6 tended to have more organic matter content at non-riparian-vegetated sites. Stream 10 showed a balanced ranking, and streams 4, 7, 8 and 9 tended to have more organic matter content at the riparian-vegetated sites. Only streams 4 and 8 showed a significant difference between riparian vegetation and no riparian vegetation (Mann–Whitney U test, $n = 10$).

Table 3.3: Comparison of organic matter data between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U Test	Exact Sig. [2*(1-tailed Sig.)]	Mean rank S1	Mean rank S2
Stream 3 (Pa)	0.151	4	7
Stream 4 (Pa)	0.008	8	3
Stream 5 (Cr)	0.056	3.6	7.4
Stream 6 (Cr)	0.548	4.8	6.2
Stream 7 (Pa)	0.841	5.8	5.2
Stream 8 (Pa)	0.008	8	3
Stream 9 (Cr)	0.151	6.9	4.1
Stream 10 (Cr)	1.000	5.5	5.5

The results of the data separated by regions and by riparian vegetation showed that both regions tended to have higher organic matter content at the riparian-vegetated sites, but without a significant difference (*Waldviertel*: Wilcoxon signed-rank test, $p = 1.000$, $n = 8$; *Mostviertel*: Wilcoxon signed-rank test, $p = 0.197$, $n = 8$). Broader-based analysis, in which all data were compared and only separated by region, showed a significant difference between the overall organic matter content; the *Mostviertel* had higher concentrations than the *Waldviertel* (Kruskal–Wallis test, $X^2 = 7.472$, $p = 0.006$, $n = 80$).

The differentiation between flood and low-flow conditions of the *Schlattenbach* and the *Dollbach* showed generally higher organic matter content during flood conditions but without significant difference (Wilcoxon signed-rank test, p = 0.068, n = 8). Only examining the flood situations, sites with riparian vegetation tended to have higher organic matter content (Wilcoxon signed-rank test, p = 0.655, n = 4), while during low-flow conditions the results were coequal in which neither of the conditions dominated (Wilcoxon signed-rank test, p = 1.000, n = 4). Considering only the riparian-vegetated sites or only the non-riparian-vegetated sites, both tended to have higher organic matter contents during flood conditions, although these results were not statistically significant (riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4; non-riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4).

3.4 Grain Size Distribution

Grain size distributions were separated by two sediment fractions. The smaller silt and clay fraction and the larger sand fraction.

Silt and Clay

The silt and clay fractions had medians distributed from 0.06 g to 3.73 g for the riparian-vegetated sites and 0.12 g to 1.57 g for the non-riparian-vegetated sites. As shown in Figure 3.5, the riparian-vegetated sites showed slightly higher amounts of silt and clay than the non-riparian-vegetated sites. Statistical analysis showed no significant differences (Wilcoxon signed-rank test, p = 0.742, n = 16).

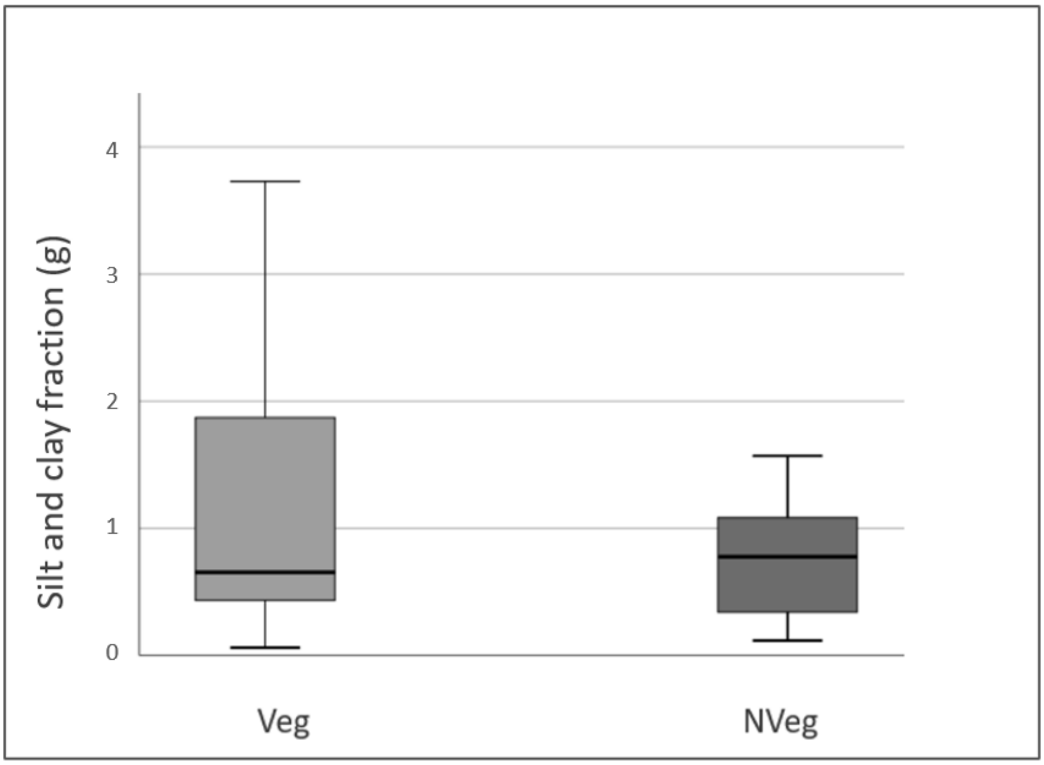

Figure 3.5: Medians of silt and clay fractions for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (*n = 16*).

The data regarding the silt and clay fractions was further analyzed by distinction of land use. The results of pasture-dominated land use tended toward higher silt and clay amounts at the riparian-vegetated sites (Wilcoxon signed-rank test, $p = 0.875$, $n = 8$). Similar to those results were the results from the crop-dominated land use, also showing more silt and clay at the riparian-vegetated sites but without significance (Wilcoxon signed-rank test, $p = 0.625$, $n = 8$).

Separation of all data only by land use and not by riparian vegetation was applied via a Kruskal–Wallis test. The results showed no significance; however, they tended toward higher silt and clay fractions at sites which had crop-dominated land use (Kruskal–Wallis test, $X^2 = 0.92$, $p = 0.338$, $n = 80$).

Results concerning the differences between riparian vegetation and its absence for each stream alone are depicted in Table 3.4. None of the streams produced significant results. Streams 3, 5, 6, 7 and 8 exhibited higher silt and clay fractions at the non-riparian-vegetated sites. Streams 4, 9 and 10 showed larger amounts of silt and clay at the riparian-vegetated sites (Mann–Whitney U test, $n = 10$).

Table 3.4: Comparison of the silt and clay fractions between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U test	Exact Sig. [2*(1-tailed Sig.)].	Mean rank S1	Mean Rank S2
Stream3 (Pa)	0.10	2	5
Stream4 (Pa)	0.10	5	2
Stream5 (Cr)	0.40	2.67	4.33
Stream6 (Cr)	0.70	3	4
Stream7 (Pa)	0.40	2.67	4.33
Stream8 (Pa)	0.40	2.67	4.33
Stream9 (Cr)	0.10	5	2
Stream10 (Cr)	1.00	3.67	3.33

A separation for regions and riparian vegetation showed that both the *Waldviertel* and the *Mostviertel* had no significant difference but slightly higher contents of silt and clay at the riparian-vegetated sites compared to the non-riparian-vegetated sites (*Waldviertel*: Wilcoxon signed-rank test, p = 0.715, n = 8; *Mostviertel*: Wilcoxon signed-rank test, p = 0.465, n = 8). A broader analysis, which only investigated the differentiation of the two regions, revealed the *Mostviertel* to have significantly higher silt and clay contents (Kruskal–Wallis test, $X^2 = 7.298$, p = 0.007, n = 48).

The analysis of the two streams that were sampled during low flow and flood situations resulted in insignificantly higher silt and clay fractions during flood conditions (Wilcoxon signed-rank test, p = 0.068, n = 8). Considering only the flood or only the low-flow conditions, both analyses demonstrated insignificantly higher amounts of silt and clay at the riparian-vegetated sites (flood: Wilcoxon signed-rank test, p = 0.180, n = 4; low flow: Wilcoxon signed-rank test, p = 0.180, n = 4). Examining all riparian-vegetated sites as well as all non-riparian-vegetated sites separately, both groups showed insignificantly larger amounts of silt and clay in flood conditions (riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4; non-riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4).

Sand

Comparison of the sand fractions of sites with and without riparian vegetation showed an insignificant difference in which the riparian-vegetated sites had lower concentrations of this sediment fraction (Wilcoxon signed-rank test, p = 0.945, n = 16) (Figure 3.6). The medians ranged from 13.86 to 56.69 g at the riparian-vegetated sites and from 9.35 to 76.51 g at the non-riparian-vegetated sites.

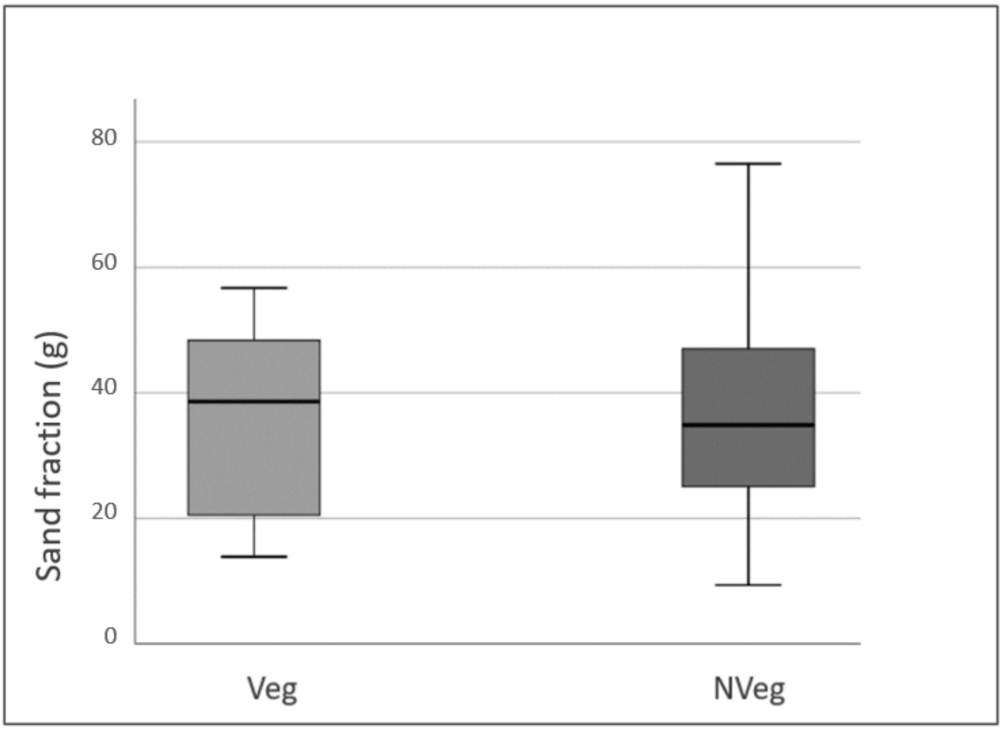

Figure 3.6: Medians of sand fractions for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (*n = 16*).

Separation of the sand fraction data by land use showed that pasture sites tended toward more sand at the non-riparian-vegetated sites, without a significant difference (Wilcoxon signed-rank test, p = 0.625, n = 8). On the contrary, crop-dominated land tended toward higher sand amounts at the riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.875, n = 8).

Further separation of the sand fraction data into land use without separating into riparian vegetation was performed via a Kruskal–Wallis test. Its results showed no significant difference but tended toward larger sand fractions at sites with crop-dominated land use (Kruskal–Wallis test, $X^2 = 2.456$, $p = 0.117$, $n = 80$).

Similar to Table 3.4, Table 3.5 shows the results of a statistical analysis which considered each stream alone and the difference between riparian vegetation and its absence. Here, regarding the sand fraction, no significant difference was observed. Streams 3, 5, 6 and 8 tended to have larger sand fractions at the non-riparian-vegetated sites and streams 4, 7, 9 and 10 tended to have larger sand fractions at the riparian-vegetated sites (Mann–Whitney U test, $n = 10$).

Table 3.5: Comparison of the sand fraction between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U test	Exact Sig. [2*(1-tailed Sig.)].	Mean rank S1	Mean Rank S2
Stream3 (Pa)	0.20	2.33	4.67
Stream4 (Pa)	0.20	4.67	2.33
Stream5 (Cr)	0.10	2	5
Stream6 (Cr)	0.70	3	4
Stream7 (Pa)	0.40	4.33	2.67
Stream8 (Pa)	0.10	2	5
Stream9 (Cr)	0.10	5	2
Stream10 (Cr)	1.00	3.67	3.33

Separation of the sand fraction data into regions and riparian vegetation revealed that both the *Waldviertel* and the *Mostviertel* had insignificantly higher sand contents at the non-riparian-vegetated sites (*Waldviertel*: Wilcoxon signed-rank test, $p = 0.273$, $n = 8$; *Mostviertel*: Wilcoxon signed-rank test, $p = 0.465$, $n = 8$). The comparison of all data, only separated by region, showed a significantly higher content of sand at the *Waldviertel* sites (Kruskal–Wallis test, $X^2 = 7.634$, $p = 0.006$, $n = 48$).

Comparing flood and low-flow conditions, sand insignificantly tended toward higher amounts during flood situations (Wilcoxon signed-rank test, $p = 0.068$, $n = 8$). Analyzing only data from flood or the low-flow conditions, both tended to show more sand at the riparian-vegetated sites (flood: Wilcoxon signed-rank test, $p = 0.180$, $n = 4$; low flow: Wilcoxon signed-rank test, $p = 0.180$, $n = 4$). The results from only the riparian-vegetated sites or only the non-riparian-vegetated sites also both tended in the same direction of higher sand amounts during flood conditions (riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4; non-riparian-vegetated: Wilcoxon signed-rank test, p = 0.180, n = 4).

3.5 Sediment Phosphorus Concentration

The sediment phosphorus concentrations were analyzed by two methods, HCl extraction for inorganic phophorus and total phosphorus.

Inorganic Phosphorus

The analysis of the HCl-extracted phosphorus (Figure 3.7) showed medians ranging from 14.76 µg g^{-1} to 1409.55 µg g^{-1} for the riparian-vegetated sites and medians from 15.22 µg g^{-1} to 1512.22 µg g^{-1} for the non-riparian-vegetated sites. The statistical methods showed no significant differences for riparian vegetation, whereas the riparian-vegetated sites had larger amounts of HCl-extractable phosphorus (Wilcoxon signed-rank test, p = 0.742, n = 16).

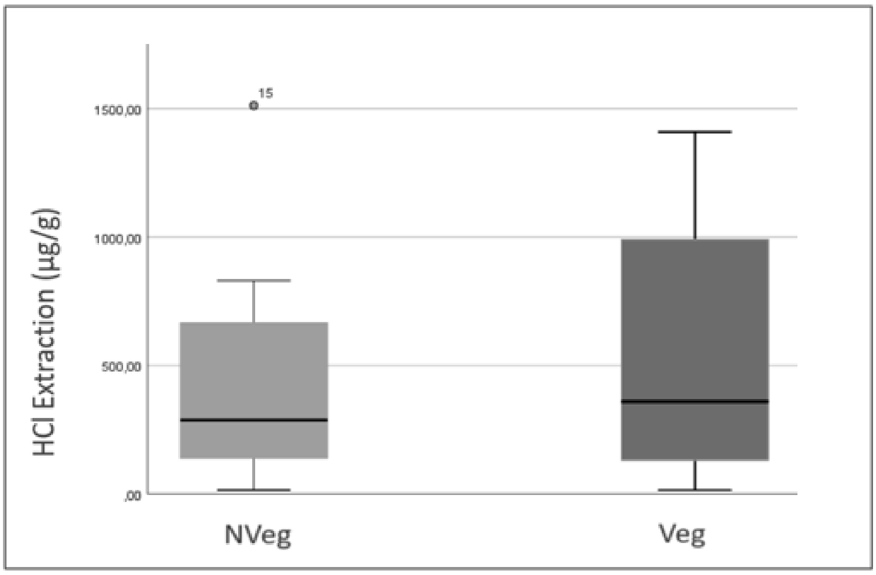

Figure 3.7: Medians of HCl extraction for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (n = *16*).

A differentiated view of the data, separated by land use, indicated diverse results. Pasture-dominated land use showed insignificantly larger amounts of HCl-extractable phosphorus at riparian-vegetated sites than at non-riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.250, n = 8). Crop-dominated land use produced results in the opposite direction, in which non-riparian-vegetated sites had larger amounts, also without significance (Wilcoxon signed-rank test, p = 0.250, n = 8).

Differentiation only by land use without separating the riparian vegetation sites showed a significant difference. Crop dominated land use had higher concentrations in sediment phosphorus that was extractable by this method (Kruskal–Wallis test, X^2 = 4.164, p = 0.041, n = 80).

Table 3.6 shows the results of each stream investigated separately for differences between presence and absence of riparian vegetation. There was no significant difference detectable for any of the streams. Nevertheless, the results show that streams 3, 4, 5, 8 and 10 had higher sediment phosphorus concentrations at the riparian-vegetated sites. Streams 6, 7 and 9 seemed contrary, showing higher concentrations at non-riparian-vegetated sites (Mann–Whitney U test, n = 10).

Table 3.6: Comparison of sediment phosphorus concentration from the HCl-extraction method between sites with riparian vegetation (S1) and sites without riparian vegetation (S2) for each stream separately via Mann–Whitney U tests. The streams were differentiated by pasture (Pa) and crop land use (Cr).

Mann–Whitney U test	**Exact Sig. [2*(1-tailed Sig.)].**	**Mean rank S1**	**Mean Rank S2**
Stream3 (Pa)	0.10	5	2
Stream4 (Pa)	0.10	5	2
Stream5 (Cr)	1.00	3.67	3.33
Stream6 (Cr)	0.40	2.67	4.33
Stream7 (Pa)	1.00	3.33	3.67
Stream8 (Pa)	0.10	5	2
Stream9 (Cr)	0.40	2.67	4.33
Stream10 (Cr)	1.00	3.67	3.33

Further analysis separated by regions and presence or absence of riparian vegetation showed diverse results. In the *Waldviertel*, results insignificantly tended toward higher concentrations at riparian-vegetated sites, while in the *Mostviertel* results insignificantly tended toward higher concentrations at the non-riparian-vegetated sites (*Waldviertel*: Wilcoxon signed-rank test, p = 0.273, n = 8; *Mostviertel*: Wilcoxon signed-rank test, p = 0.273, n = 8). A broader analysis of the data separated by region and not by riparian vegetation revealed no significant difference, although the *Waldviertel* had slightly higher concentrations (Kruskal–Wallis test, $X^2 = 6.537$, p = 0.01, n = 80).

Comparison of flood and low-flow conditions via the two streams, which were also sampled in flood conditions, showed insignificantly higher concentrations during flood conditions (Wilcoxon signed-rank test, p = 0.715, n = 8). A separation of the data into flood and low flow showed that both conditions tended toward higher concentrations at the non-riparian-vegetated sites (flood: Wilcoxon signed-rank test, p = 0.655, n = 4; low flow: Wilcoxon signed-rank test, p = 0.180, n = 4). Another separation by presence or absence of riparian vegetation resulted in insignificant but diverse results. The riparian-vegetated sites tended to have higher concentrations during flood scenarios (Wilcoxon signed-rank test, p = 0.665, n = 4), and the non-riparian-vegetated sites tended to have higher concentrations during low-flow conditions (Wilcoxon signed-rank test, p = 0.180, n = 4).

Total Phosphorus

Results for total phosphorus were unavailable due to technical difficulties of the method. In many streams, the inorganic fraction exceeded the total phosphorus, resulting in an unrealistic estimate. As the organic phosphorus fraction is the difference between the total and the inorganic phosphorus fractions, neither the total nor the organic phosphorus fraction was further considered in this thesis.

3.6 Water Nutrients

Water-column SRP concentrations ranged from 8.3 µg L^{-1} to 67 µg L^{-1} for riparian-vegetated sites and from 9.8 µg L^{-1} to 69 µg L^{-1} for non-riparian-vegetated sites. The applied statistical methods could not detect significant differences; however, higher SRP concentrations were observed at the non-riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.123, n = 16). These results are visualized in Figure 3.8.

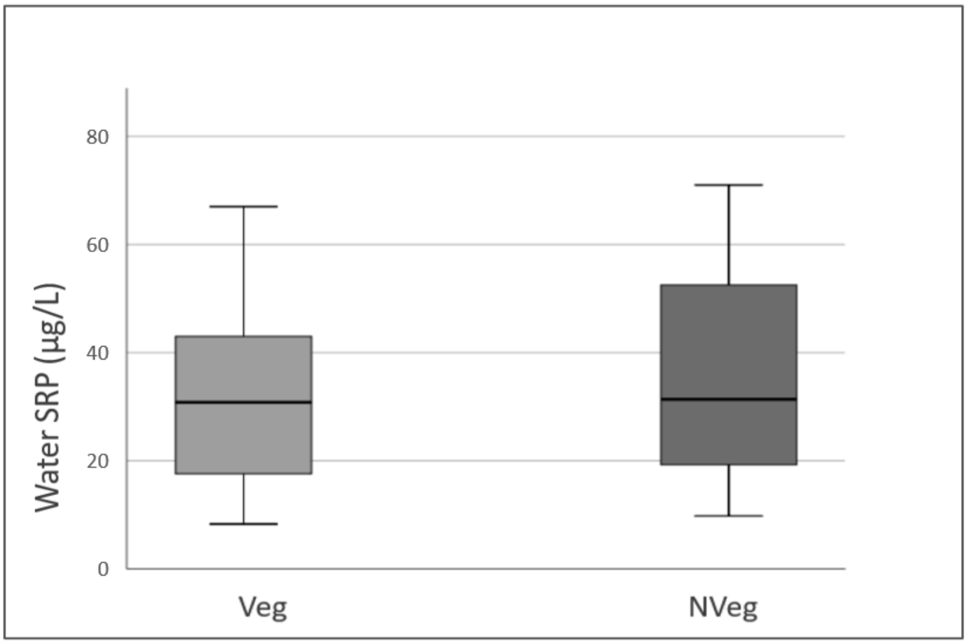

Figure 3.8: Water SRP concentrations for all sites with riparian vegetation (Veg) and without riparian vegetation (NVeg). Shown are the maximal and minimal values, 25 % and 75 % quantiles as well as the medians (*n = 16*).

The results of further differentiation, not only by riparian vegetation but also by land use, followed the same course. The pasture sites exhibited a tendency toward higher concentrations of SRP at non-riparian-vegetated sites, but the difference was not significant (Wilcoxon signed-rank test, p = 0.273, n = 8). The same results occurred for the cropland sites (Wilcoxon signed-rank test, p = 0.273, n = 8).

A Kruskal–Wallis test was applied to detect general differences between the two land use types without separating them by riparian vegetation. The results showed significantly higher SRP concentrations in crop-surrounded streams compared to pasture-surrounded streams (Kruskal–Wallis test, X^2 = 6.353, p = 0.012, n = 16).

Further separation of the data by region (*Mostviertel* or *Waldviertel*) as well as riparian vegetation showed diverse results. In the *Waldviertel* there was no significant difference and no detectable tendency (Wilcoxon signed-rank test, p = 1.000, n = 8). The *Mostviertel* demonstrated results which had an insignificant tendency toward higher SRP concentrations at the non-riparian-vegetated sites (Wilcoxon signed-rank test, p = 0.068, n = 8). Additionally, a Kruskal–Wallis test was applied to examine differences between the regions without separating the data by riparian vegetation. There was no difference or tendency shown by the results (Kruskal–Wallis test, X^2 = 0.000, p = 1.000, n = 16).

3.7 Summary of the Results

In Table 3.7, the results concerning the presence and absence of riparian vegetation are summarized together. As the main parameters were confronted with each stream or different coalitions of streams, some tendencies and interactions became visible.

Table 3.7: Summary of all result indications. The first line divides into parameters while the first column divides into streams and stream aggregations. The content of the table indicates whether the data values from riparian-vegetated sites (Veg) or non-riparian-vegetated sites (NVeg) are greater. The underlined content determines results with significant differences, and the rectangles indicate accordance with the hypothesis.

	EPCo (µg/L)	**Respiration (mg/g h)**	**Silt and Clay (g)**	**P-HCl (µg/g)**	**Organ. M. (g)**
All streams	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg
Pasture streams	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg
Cropland streams	Veg < NVeg	Veg > NVeg	Veg > NVeg	Veg < NVeg	Veg < NVeg
Stream 3 (Pasture)	Veg > NVeg	Veg > NVeg	Veg < NVeg	Veg > NVeg	Veg < NVeg
Stream 4 (Pasture)	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg	Veg > NVeg
Stream 5 (Crop)	Veg < NVeg	Veg < NVeg	Veg < NVeg	Veg > NVeg	Veg < NVeg
Stream 6 (Crop)	Veg < NVeg	Veg > NVeg	Veg < NVeg	Veg < NVeg	Veg < NVeg
Stream 7 (Pasture)	Veg > NVeg	Veg > NVeg	Veg < NVeg	Veg < NVeg	Veg > NVeg
Stream 8 (Pasture)	Veg < NVeg	Veg < NVeg	Veg < NVeg	Veg > NVeg	Veg > NVeg
Stream 9 (Crop)	Veg < NVeg	Veg < NVeg	Veg > NVeg	Veg < NVeg	Veg > NVeg
Stream 10 (Crop)	Veg<NVeg	Veg < NVeg	Veg > NVeg	Veg > NVeg	Veg = NVeg

4. Discussion

4.1 Phosphorus Buffer Potential

In contrast to the first hypothesis, the EPC_0 did not differ between sites with and without riparian vegetation. The average EPC_0 was 40.3 µgL^{-1} at sites with riparian vegetation and 38.6 µgL^{-1} at sites without riparian vegetation. Other studies have shown similar values, which were not differentiated by the extent of the stream's riparian vegetation (McDowell et al., 2002; McDaniel et al., 2009; Teufl et al., 2012). The first hypothesis intended to determine whether 1 km of stream length with riparian vegetation can result in significantly different phosphorus buffering capabilities of the stream sediments. It was expected that the riparian vegetation would limit the nutrient inputs, which then would result in lower EPC_0 and thus higher phosphorus buffering capacities. Weigelhofer et al. (2012) differentiated headwater streams with and without riparian buffer strips in a study regarding in-stream nutrient retention in a neighbouring region with similar agricultural streams. The study did not show significant differences between sites with and without riparian buffers, concerning in-stream phosphorus retention. This implies that in-stream nutrient retention cannot balance excess nutrient loads and that riparian buffers are needed to reduce these loads (Weigelhofer et al., 2012).

Reasons for the lack of riparian effects on the EPC_0 might be found in the grain size distribution, as particle size influences the phosphorus adsorption capacity of the sediments (Clarendon et al., 2019; McDaniel et al., 2009; see chapter 1.3). The third hypothesis, in which higher amounts of small sediment grain size were expected to be found in non-riparian-vegetated sites was not confirmed. An insignificant tendency toward higher quantities of smaller grain sizes in riparian-vegetated sites was detected. As the phosphorus uptake capacity of sediments rises with decreasing sediment size (Klotz, 1988; Munn and Meyer, 1990; McDaniel et al., 2009), the contradictions between the first hypothesis and the results can be explained. The reason that streams with riparian vegetation might have larger amounts of small sediment grain sizes originates in structural heterogeneity. More roots and debris, as well as diversity in the riverbed structure, lead to more possibilities for fine sediment to be trapped in that area (Ensign and Doyle, 2005). In the study sites of this thesis, fine sediments could enter upstream through the non-riparian-vegetated area and would be trapped within the riparian-vegetated areas, resulting in higher $EPC_{0.}$ It is conceivable that the study-site length of 1 km was too short to see the effects of the riparian buffers. This emphasizes the importance of sufficient buffer length and consistencies throughout the entire stream to be efficient in buffering nutrient inputs (Weigelhofer, 2016; Weigelhofer et al., 2022). Even small bypasses can interfere with the purpose of riparian buffers. In these conditions, nutrient loads can enter the water column unbuffered via soil erosion and thus diminish the purpose of riparian buffers (Leeds-Harrison et al., 1999; Corell, 2005).

Another potential cause for the EPC_0 results may lie within the microbial communities. Although their influence on the phosphorus buffer capacity of stream sediments differs among studies (DiCarlo et al., 2020), microbial communities certainly benefit from phosphorus within the sediments (Lottig and Stanley, 2007; House, 2003). According to Lottig and Stanley (2007), there is a shift from abiotically to biotically dominated buffering capacity with rising sediment grain size (see Figure 1.1). Thus, the biotic role in the buffer capacity of the present study sites may have been rather small (Lottig and Stanley, 2007; McDaniel et al., 2009). In any case, these microbial communities could have altered the results of EPC_0, as they would assimilate phosphorus (Stutter and Lumsdon, 2008) and thereby alter the final SRP concentration of the EPC_0 method (see Subsection 2.3.1). In theory, this could explain the lack of a difference in EPC_0 values between vegetated and unvegetated sites. Concerning algae, their autotrophic activity did not play a crucial role within this study, as the sediment samples were sieved (see Subsection 2.2.1). Heterotrophs, however, drive the phosphorus cycle in streams with limited light penetration (Pusch et al., 1998), suggesting greater assimilations in sediments of riparian vegetation. On the other hand, organic matter provided by riparian vegetation is not as bioavailable as organic matter provided by algae, which thrive better in sunlight (Reddy et al., 1999; Withers and Jarvie, 2008). These contradicting arguments and the circumstance that biotic uptake was not analyzed within this study have led to the assumption that the microbial community did not affect the results in this thesis.

Comparison between water SRP and EPC_0 can indicate whether sites are potential phosphorus sinks or sources (Jarvie et al., 2005). Figure 3.2 reveals that most of the sites had higher EPC_0 than SRP concentrations and thus were acting as potential phosphorus sources. Other authors have reported similar results regarding streams of lower orders (McDaniel et al., 2009). Another study found that stream sediments switched from source to sink above a threshold of approximately 90 µg L^{-1} to 150 µg L^{-1} (Weigelhofer et al., 2018). This thesis, however, could not confirm this shift. There is a slight curve visible in Figure 3.2; nevertheless, all sites were acting as potential sources, and the highest EPC_0 median (158.7 µg L^{-1}) also appeared to be the largest potential source of phosphorus. The reasons for this switch could have originated from either phosphorus saturation or a disrupted hyporheic exchange (Stutter and Lumsdon, 2008). The ability of sediments to act as a phosphorus source can be activated by two circumstances. One is the mobilization of sediments (Smith et al., 2006), which is unlikely to have occurred in this study, as the sampling was conducted in baseflow conditions. The second is reduced SRP in the water column (Weigelhofer et al., 2018). The latter seems to be the case in this study and is largely dependent on the quantity and timing of water flowing through the hydrologic system (Choquette et al., 2019).

As most of the sites acted as a phosphorus source, there was no significant difference detected between riparian-vegetated sites and non-riparian-vegetated sites. This lack of a difference is consistent with the findings regarding EPC_0 and grain size distribution. Assignment as a sink or a source was dependent on the results of the SRP measurements in addition to those for EPC_0 (see Section 3.6). Concentrations ranged from 8.3 µg L^{-1} to 69 µg L^{-1}. Comparisons to a stream catchment from the eastern USA show very similar concentrations (McDowell et al., 2003; Lotting and Stanley, 2007), while other authors with results from Lower Austria have reported much higher concentrations, ranging from 2.3 µg L^{-1} to 407.9 µg L^{-1} (Weigelhofer et al., 2018).

The study design of this thesis differentiated between two land use types (see Section 2.1). Half of the included streams had a dominant surrounding land use of crop fields, and the other half were used as pastures. In Hypothesis 4 (see Section 1.8), vegetated buffer strips were expected to show greater effects in cropland than in pasture-dominated land because the riparian vegetation retains soil particles eroded from bare soils. The results showed no significant difference in either direction; however, all cropland streams showed the tendency toward higher EPC_0 at non-riparian-vegetated sites, and most pasture streams tended to show higher values at the riparian-vegetated sites. A possible explanation might lie in the different fertilization strategies associated with land use. As croplands are fertilized more intensively (Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, 2017), larger amounts of phosphorus are generated and transported to the streams. Those larger amounts may increase the effects of riparian vegetation, which would increase the differences observed between riparian vegetation and lack of riparian vegetation. The results of this thesis would support that assumption, as comparisons between all crop sites and all pasture sites showed significantly higher values of EPC_0, SRP and inorganic phosphorus in the sediments at crop sites. This trend has also been observed in other studies (Weigelhofer et al., 2018; McDowell et al., 2002).

An additional possibility for the different circumstances between crop fields and pastures could be related to the discharge of the stream. The median discharge of the cropland streams was much lower than those of the pasture streams (Table 2.1). This ensured more possibilities to retain phosphorus in the streams surrounded by crop fields, as there is more streambed surface per discharge in those streams (Jarvie et al., 2005).

4.2 Respiration

The average respiration was 0.25 mg g^{-1} h^{-1} at vegetated sites and 0.23 mg g^{-1} h^{-1} at unvegetated sites, further demonstrating the lack of a difference between the sites. Compared to other studies, the level of the sediment respiration was low but still reasonable (Hill et al., 2000; Pastor et al., 2017; Laperriere et al., 2020). In stream ecosystems, the oxygen consumption of microorganisms and chemical reactions which reduce oxygen (Sø et al., 2022) stand opposed to the autotrophic production of oxygen and the introduction of oxygen by hydrologic turbulence (Correa-González et al., 2014). However, the respiration measurements were conducted in darkness and the sediment samples were sieved; hence, autotrophs did not play a crucial role (see Subsection 2.3.2). This context led to the assumption that respiration showed the sediment's metabolism, in turn indicating the intensity of the nutrient assimilation (Staehr et al., 2012).

The sites without riparian vegetation were expected to contain larger amounts of organic matter, as the autochthonous production of organic matter benefits from larger nutrient availabilities (Webster et al., 1995; Hagen et al., 2010). Sediment respiration, in turn, increases with organic matter (Cole et al., 1988; Hedin 1990; Ferreira et al., 2020). Consequently, according to the second hypothesis, the sites without riparian buffers were expected to show more respiration than those with such buffers. However, this assumption became negligible, as the results presented in Sections 3.3, 3.5 and 3.6 showed a lack of difference in levels of organic matter, sediment phosphorus and water SRP between the sites. A possible reason for the lack of a difference in organic matter between the sites is that organic matter is not only promoted by autochthonous production in unvegetated sites but also by inputs such as leaf litter in vegetated sites (Webster et al., 1995). Therefore, the respiration results do not suggest a significant influence of the microbial community on the phosphorus buffer potential between the sites. The results reflect a consistent lack of a difference among all parameters regarding riparian vegetation. This observation leads back to the question of whether the study-site length of 1 km with riparian vegetation was sufficient to detect a significant difference.

In the fourth hypothesis, the effects of Hypotheses 1, 2 and 3 were expected to be greater in crop-dominated land than in pastures. However, the results showed no differences between presence or absence of riparian vegetation concerning EPC_0, sediment respiration, sediment grain size, organic matter and sediment phosphorus in both types of land use (see Table 3.7). The only significant differences were detectable if land use was compared without differentiating by riparian vegetation. In that case, EPC_0, sediment phosphorus and water SRP had higher amounts in crop land use, but respiration and organic matter were not different. As mentioned previously, respiration is connected to organic matter (Cole et al., 1988; Hedin 1990; Ferreira et al., 2020), which can explain the lack of a difference in respiration between cropland and pastures. Nevertheless, the indifference of organic matter between crop and pasture seems inconsequential, as other authors have concluded that the amount of organic matter is dependent on land use (Wilson and Xenopoulos, 2009; Weigelhofer et al., 2018; Shang et al., 2018; Pisani et al., 2020).

4.3 Regions

Half of the streams analyzed in this study are positioned in the *Waldviertel* and half in the *Mostvietel*. Disparities in the effects of the presence or absence of riparian vegetation on EPC_0, respiration, sediment grain size, organic matter and sediment nutrient contents were expected to be greater in the *Mostviertel* than in the *Waldviertel*. The assumption of the hypothesis was made because the streams of the *Mostviertel*, especially the *Schlattenbach* and the *Dollbach*, are situated in the highly agriculturally utilized area just to the south of the Danube River. The only comparable stream in the *Waldviertel* is the *Große Taffa* (AGES, 2018). The higher potential nutrient inputs in these areas were expected to increase the differences between sites with riparian vegetation and sites without riparian vegetation and thereby show larger differences in the *Mostviertel* than in the *Waldviertel*.

A comparison between the regions without separation by riparian vegetation shows that the *Waldviertel* had a significantly larger sand fraction and the *Mostviertel* a significantly larger organic matter and silt and clay fraction. These findings would partly promote the hypothesis, as smaller sediment sizes increase the EPC_0 (Lottig and Stanley, 2007) and consequently, would lead to larger differences between the sites of the *Mostviertel*. The results did not confirm that hypothesis, as both the *Waldviertel* and the *Mostviertel* showed no significant tendencies between sites with riparian vegetation and sites without. As mentioned in Section 4.1, other studies have found similar results concerning riparian buffers (Weigelhofer et al., 2012). The only indices promoting the hypothesis are found if looking at the streams separately. Streams 8 and 9 are the only streams which show significantly higher EPC_0 results in the non-riparian-vegetated sites and both are streams of the *Mostviertel* (see table 4). But other than that, the lack of difference between the sites and regions, might also be explained with arguments which were already presented in section 4.1. Specifically, the distribution of fine sediments between riparian-vegetated and non-riparian-vegetated sites and the severity of bypasses in buffer strips.

4.4 Floods

Hypothesis 5 states that the effects of the first three hypotheses are all indifferent to flood conditions. These effects concerned the difference between areas with riparian vegetation and those without riparian vegetation for the parameters of phosphorus retention capacities, respiration, sediment grain size, sediment nutrients and organic matter. The hypothesis was confirmed by the results, as all these parameters showed no significant difference between areas with riparian vegetation and areas without it. The hypothesis expected the increased discharge to blend any differences between the consecutive sites. As higher discharges increase the stream length, nutrients need to be retained again in biotic or abiotic ways (Covino, 2016). Floods with access to floodplains can lead to nutrient trapping and other beneficial occurrences (Weigelhofer et al., 2018). But, the sites of this thesis were all within agricultural streams which are not connected to floodplains. Consequently, floods rather led to the introduction of more nutrients to the system through increased runoff (Weigelhofer et al., 2018; Ockenden, 2012; Banner et al., 2009). However, there was no significant difference detectable between the flood conditions and the low-flow conditions.

4.5 Conclusions

In this study, eight streams were investigated regarding the phosphorus retention characteristics of their benthic sediments. All streams consisted of a section without riparian vegetation which was consecutively followed by a section with riparian vegetation. The riparian-vegetated sites were expected to buffer nutrient inputs and thereby increase the sediment's ability to retain phosphorus. However, the results did not show significant differences between the sites concerning all parameters. These findings suggest that 1 km of buffered stream length is not sufficient to make a detectable difference. The reason behind these results seems to lie in the grain size distribution of the sediments, as this parameter affects the ability of the sediments to retain phosphorus. This underlines the importance of including hydrodynamic sediment transport in stream management. Moreover, these findings highlight the importance of continuous riparian vegetation without bypasses. The study design differentiated between cropland and pasture use of the surrounding areas. It was expected that crop sites, as they have higher nutrient loads, would demonstrate a larger difference between the sites with and without riparian vegetation. This effect was not reflected in the results, but there were predictable and significantly higher nutrient loads in the collective crop sites. This again emphasizes the importance of continuous riparian vegetation, especially in nutrient-intensive areas. Unfortunately, in those areas, the immediate surroundings of the streams are often highly agriculturally utilized. The implementation of buffer strips represents opportunity costs that many are not able or willing to sustain. In this case, agro-economic efficiency stands against the many, often unnoticed, benefits that riparian buffers provide. Besides their nutrient buffer effects, the stabilization of stream banks and water temperature, flood mitigation and habitat provisioning represent a further variety of societal benefits.

To summarize, it is essential to expand continuously connected riparian buffers to increase the resilience of streams. The ecological and socio-economic importance of proper basin-wide nutrient management practices becomes tangible when considering historical algal explosions and their consequences. The world's ever-increasing climate change requires measures which are basin-wide and interdisciplinary. To find the best measures, further investigations are needed. Some authors have already provided well-considered suggestions concerning the implementation of efficient buffer strips (Ramler et al., 2022), but the political response remains unseen.

A useful follow-up study would involve further analysis of the efficiency of buffer strips to determine what kind of vegetation in which areas and climates are most fitting to increase the benefits and mitigate flooding, erosion and rising temperatures. Furthermore, it is essential to optimize implementation processes and to identify the best incentives for landowners to alter their relationship with adjacent water bodies.

Citations:

AGES, 2018. ASOC, Österreichische Karte des organischen Bodenkohlenstoffs, Endbericht zum Forschungsprojekt Nr. 101255. Available at: Available at: <https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&cad=rja&uact=8&ved=2ahUKEwjTnoXDjozuAhVK-qQKHZjHCCgQFjAAegQIARAC&url=https%3A%2F%2Fwww.dafne.at%2Fdafne_plus_homepage%2Fdownload.php%3Ft%3DProjectReportAttachment%26k%3D4428&usg=AOvVaw2mHAx_CAJxkvGTK9GGBjBB> [Accessed 8 January 2021].

Allan, J., 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, 35(1), pp.257-284.

Banner, E., Stahl, A. and Dodds, W., 2009. Stream Discharge and Riparian Land Use Influence In-Stream Concentrations and Loads of Phosphorus from Central Plains Watersheds. *Environmental Management*, 44(3), pp.552-565.

Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Richtlinien für die Sachgerechte Düngung, 7. Auflage. Available at: https://info.bml.gv.at/themen/landwirtschaft/landwirtschaft-in-oesterreich/bodenschutz/bodenschutz-duengung/stickstoff-und-phosphorbilanz,-umstellung-auf-eurostatvorgaben.html> [Accessed 8 January 2021].

Cao, X., Song, C., Xiao, J. and Zhou, Y. (2018). The Optimal Width and Mechanism of Riparian Buffers for Storm Water Nutrient Removal in the Chinese Eutrophic Lake Chaohu Watershed. *Water, 10*(10), 1489. doi: 10.3390/w10101489

Choquette, A., Hirsch, R., Murphy, J., Johnson, L. and Confesor, R., 2019. Tracking changes in nutrient delivery to western Lake Erie: Approaches to compensate for variability and trends in streamflow. *Journal of Great Lakes Research*, 45(1), pp.21-39.

Clarendon, S., Weaver, D., Davies, P. and Coles, N., 2019. The influence of particle size and mineralogy on both phosphorus retention and release by streambed sediments. *Journal of Soils and Sediments*, 19(5), pp.2624-2633.

Cole, J. J., Findlay, S. and Pace, M. L., 1988. Bacterial production in fresh and saltwater ecosystems: a cross-system overview. *Marine ecology progress series. Oldendorf*, *43*(1), 1-10.

Correa-González, J. C., del Carmen Chávez-Parga, M., Cortés, J. A. and Pérez-Munguía, R. M., 2014. Photosynthesis, respiration and reaeration in a stream with complex dissolved oxygen pattern and temperature dependence. *Ecological modelling*, *273*, 220-227.

Correll, D., 2005. Principles of planning and establishment of buffer zones. *Ecological Engineering*, 24(5), pp.433-439. Correll, D., 2005. Principles of planning and establishment of buffer zones. *Ecological Engineering*, 24(5), pp.433-439.*ƒ*

Covino, T., 2017. Hydrologic connectivity as a framework for understanding biogeochemical flux through watersheds and along fluvial networks. *Geomorphology*, 277, pp.133-144.

DiCarlo, A., Weisener, C. and Drouillard, K., 2020. Evidence for Microbial Community Effect on Sediment Equilibrium Phosphorus Concentration (EPC0). *Bulletin of Environmental Contamination and Toxicology*, 105(5), pp.736-741.

Edmondson, W. and Vollenweider, R. A. 1968. Water management research. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors in eutrophication. Organization for Economic Co-operation and De. *Limnology and Oceanography*, 15(1), pp.169-170.

Ensign, S. and Doyle, M. (2005). In-channel transient storage and associated nutrient retention: Evidence from experimental manipulations. *Limnology And Oceanography*, *50*(6), 1740-1751. doi: 10.4319/lo.2005.50.6.1740

Ferreira, V., Elosegi, A., D Tiegs, S., von Schiller, D. and Young, R., 2020. Organic matter decomposition and ecosystem metabolism as tools to assess the functional integrity of streams and rivers–a systematic review. *Water*, *12*(12), 3523.

Froelich, P., 1988. Kinetic control of dissolved phosphate in natural rivers and estuaries: A primer on the phosphate buffer mechanism. *Limnology and Oceanography*, 33(4_part_2), pp.649-668.

Geologische Bundesanstalt. Geologische Karte 1:2.000.000. Available at: <https://www.geologie.ac.at/forschung-entwicklung/kartierung-landesaufnahme/geologie/geologische-karte-12000000> [Accessed 7 January 2021].

Hagen, E. M., McTammany, M. E., Webster, J. R. and Benfield, E. F. 2010. Shifts in allochthonous input and autochthonous production in streams along an agricultural land-use gradient. *Hydrobiologia*, *655*(1), 61-77.

Hedin, L. O., 1990. Factors controlling sediment community respiration in woodland stream ecosystems. *Oikos*, 94-105.

Hill, B., Hall, R., Husby, P., Herlihy, A. and Dunne, M., 2000. Interregional comparisons of sediment microbial respiration in streams. *Freshwater Biology*, 44(2), pp.213-222.

House, W., 2003. Geochemical cycling of phosphorus in rivers. *Applied Geochemistry*, 18(5), pp.739-748.

Jarvie, H., Sharpley, A., Withers, P., Scott, J., Stutter, B. and Neal, C., 2013. Phosphorus Mitigation to Control River Eutrophication: Murky Waters, Inconvenient Truths, and "Postnormal" Science. *Journal of Environmental Quality*, 42(2), pp.295-304.

Jarvie, H., Jürgens, M., Williams, R., Neal, C., Davies, J., Barrett, C. and White, J., 2005. Role of river bed sediments as sources and sinks of phosphorus across two major eutrophic UK river basins: the Hampshire Avon and Herefordshire Wye. *Journal of Hydrology*, 304(1-4), pp.51-74.

Klik, A., 2004. Bodenerosion durch Wasser. Ländlicher Raum 2004, 6, 1–11.

Klotz, R., 1988. Sediment Control of Soluble Reactive Phosphorus in Hoxie Gorge Creek, New York. *Canadian Journal of Fisheries and Aquatic Sciences*, 45(11), pp.2026-2034.

Laperriere, S., Hilderbrand, R., Keller, S., Trott, R. and Santoro, A., 2020. Headwater Stream Microbial Diversity and Function across Agricultural and Urban Land Use Gradients. *Applied and Environmental Microbiology*, 86(11).

Leeds-Harrison, P., Quinton, J., Walker, M., Sanders, C. and Harrod, T. (1999). Grassed buffer strips for the control of nitrate leaching to surface waters in headwater catchments. *Ecological Engineering*, *12*(3-4), 299-313. doi: 10.1016/s0925-8574(98)00075-5

Lottig, N.R. and Stanley, E.H., 2007. Benthic sediment influence on dissolved phosphorus concentrations in a headwater stream. Biogeochemistry 84:297–309.

Macrae, M., Zhang, Z., Stone, M., Price, J., Bourbonniere, R. and Leach, M., 2011. Subsurface Mobilization of Phosphorus in an Agricultural Riparian Zone in Response to Flooding from an Upstream Reservoir. *Canadian Water Resources Journal / Revue canadienne des ressources hydriques*, 36(4), pp.293-311.

McDaniel, M., David, M. and Royer, T., 2009. Relationships between Benthic Sediments and Water Column Phosphorus in Illinois Streams. *Journal of Environmental Quality*, 38(2), pp.607-617.

McDowell, R., Sharpley, A. and Chalmers, A. (2002). Land use and flow regime effects on phosphorus chemical dynamics in the fluvial sediment of the Winooski River, Vermont. *Ecological Engineering*, *18*(4), 477-487. doi: 10.1016/s0925-8574(01)00108-2

McDowell, R., Sharpley, A. and Folmar, G., 2003. Modification of phosphorus export from an eastern USA catchment by fluvial sediment and phosphorus inputs. *Agriculture, Ecosystems & Environment*, 99(1-3), pp.187-199.

Malá, J. and Lagová, M., 2014. Comparison of digestion methods for determination of total phosphorus in river sediments. *Chemical Papers*, 68(8).

Munn, N. and Meyer, J., 1990. Habitat-Specific Solute Retention in Two Small Streams: An Intersite Comparison. *Ecology*, 71(6), pp.2069-2082.

NÖ Agrarbezirksbehörde. Unser Boden, Bodentypen in Niederösterreich, 2016. Available at: <http://www.unserboden.at/750-0-Soilmap-News.htm?&goback=51> [Accessed 8 January 2021].

Naegeli, M. and Uehlinger, U., 1997. Contribution of the Hyporheic Zone to Ecosystem Metabolism in a Prealpine Gravel-Bed-River. *Journal of the North American Benthological Society*, 16(4), pp.794-804.

Nogaro, G., Datry, T., Mermillod-Blondin, F., Descloux, S. and Montuelle, B., 2010. Influence of streambed sediment clogging on microbial processes in the hyporheic zone. *Freshwater Biology*, 55(6), pp.1288-1302.

Ockenden, M., Deasy, C., Quinton, J., Bailey, A., Surridge, B. and Stoate, C., 2012. Evaluation of field wetlands for mitigation of diffuse pollution from agriculture: Sediment retention, cost and effectiveness. *Environmental Science & Policy*, 24, pp.110-119.

OESTERREICHISCHES NORMUNGSINSTITUT, ÖNORM EN ISO 14688-1: 2013 15 11; Geotechnische Erkundung und Untersuchung - Benennung, Beschreibung und Klassifizierung von Boden - Teil 1: Benennung und Beschreibung (ISO 14688-1:2002).

Osborne, L. and Kovacic, D., 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology*, 29(2), pp.243-258.

Pastor, A., Lupon, A., Gómez-Gener, L., Rodríguez-Castillo, T., Abril, M., Arce, M., Aristi, I., Arroita, M., Bravo, A., Catalán, N., del Campo, R., Casas-Ruiz, J., Estévez, E., Fernández, D., Fillol, M., Flores, L., Freixa, A., Gimenez Grau, P., González-Ferreras, A., Catalán, N., 2017. Local and regional drivers of headstream metabolism: insights from the first AIL collaborative project. Limnetica. 36. 1-38. 10.23818/limn.36.06.

Pisani, O., Bosch, D. D., Coffin, A. W., Endale, D. M., Liebert, D. and Strickland, T. C., 2020. Riparian land cover and hydrology influence stream dissolved organic matter composition in an agricultural watershed. *Science of The Total Environment*, *717*, 137165.

Poeppl, Dilly, Haselberger, Renschler and Baartman, 2019. Combining Soil Erosion Modeling with Connectivity Analyses to Assess Lateral Fine Sediment Input into Agricultural Streams. *Water*, 11(9), p.1793.

Pusch, M., Fiebig, D., Brettar, I., Eisenmann, H., Ellis, B. and Kaplan, L. et al. (1998). The role of micro-organisms in the ecological connectivity of running waters. *Freshwater Biology*, *40*(3), 453-495. doi: 10.1046/j.1365-2427.1998.00372.x

Ramler, D., Stutter, M., Weigelhofer, G., Quinton, J., Hood-Nowotny, R. and Strauss, P., 2022. Keeping Up with Phosphorus Dynamics: Overdue Conceptual Changes in Vegetative Filter Strip Research and Management. *Frontiers in Environmental Science*, 10.

Ruban, V., López-Sánchez, J., Pardo, P., Rauret, G., Muntau, H. and Quevauviller, P., 2001. Harmonized protocol and certified reference material for the determination of extractable contents of phosphorus in freshwater sediments - A synthesis of recent works. *Fresenius' Journal of Analytical Chemistry*, 370(2-3), pp.224-228.

Reddy, K., Kadlec, R., Flaig, E. and Gale, P., 1999. Phosphorus Retention in Streams and Wetlands: A Review. *Critical Reviews in Environmental Science and Technology*, 29(1), pp.83-146.

Shang, P., Lu, Y., Du, Y., Jaffé, R., Findlay, R. H. and Wynn, A., 2018. Climatic and watershed controls of dissolved organic matter variation in streams across a gradient of agricultural land use. *Science of the Total Environment*, *612*, 1442-1453.

Simpson, Z., McDowell, R., Condron, L., McDaniel, M., Jarvie, H. and Abell, J., 2021. Sediment phosphorus buffering in streams at baseflow: A meta-analysis. *Journal of Environmental Quality*, 50(2), pp.287-311.

Schindler, D., 2006. Recent advances in the understanding and management of eutrophication. *Limnology and Oceanography*, 51(1part2), pp.356-363.

Smith, D., Warnemuende, E., Haggard, B. and Huang, C., 2006. Changes in sediment–water column phosphorus interactions following sediment disturbance. *Ecological Engineering*, 27(1), pp.71-78.

Sø, J.S., Kragh, T., Sand-Jensen, K. and Martinsen, K. T., 2022. Environmental drivers and sources of stream oxygen consumption in an agricultural lake catchment. Ecological Engineering, Volume 176

Staehr, P. A., Testa, J. M., Kemp, W. M., Cole, J. J., Sand-Jensen, K. and Smith, S. V., 2012. The metabolism of aquatic ecosystems: history, applications, and future challenges. *Aquatic Sciences*, *74*(1), 15-29.

Stutter, M., Chardon, W. and Kronvang, B., 2012. Riparian Buffer Strips as a Multifunctional Management Tool in Agricultural Landscapes: Introduction. *Journal of Environmental Quality*, 41(2), pp.297-303.

Stutter, M., Graeber, D., Evans, C., Wade, A. and Withers, P., 2018. Balancing macronutrient stoichiometry to alleviate eutrophication. *Science of The Total Environment*, 634, pp.439-447.

Stutter, M.I. and Lumsdon, D.G., 2008. Interactions of land use and dynamic river conditions on sorption equilibria between benthic sediments and river soluble reactive phosphorus concentrations. Water Res. 42:4249–4260. https://doi.org/10.1016/j. watres.2008.06.017.

Sweeney, B., Bott, T., Jackson, J., Kaplan, L., Newbold, J., Standley, L., Hession, W. and Horwitz, R., 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proceedings of the National Academy of Sciences, 101(39), pp.14132-14137.*

Teufl, B., Weigelhofer, G., Fuchsberger, J. and Hein, T., 2012. Effects of hydromorphology and riparian vegetation on the sediment quality of agricultural low-order streams: consequences for stream restoration. *Environmental Science and Pollution Research*, 20(3), pp.1781-1793.

Warkentin, M., Freese, H., Karsten, U. and Schumann, R., 2007. New and Fast Method To Quantify Respiration Rates of Bacterial and Plankton Communities in Freshwater Ecosystems by Using Optical Oxygen Sensor Spots. APPLIED AND ENVIRONMENTAL MICROBIOLOGY, Vol. 73, No. 21, pp. 6722–6729.

Webster, J. R., Wallace, J. B. and Benfield, E. F. 1995. Organic processes in streams of the eastern United States. *River stream Ecosyst. world*, 117-187.

Weigelhofer, G., Ramião, J., Pitzl, B., Bondar-Kunze, E. and O'Keeffe, J., 2018. Decoupled water-sediment interactions restrict the phosphorus buffer mechanism in agricultural streams. *Science of The Total Environment*, 628-629, pp.44-52.

Weigelhofer, G., Fuchsberger, J., Teufl, B., Welti, N. and Hein, T., 2012. Effects of Riparian Forest Buffers on In-Stream Nutrient Retention in Agricultural Catchments. *Journal of Environmental Quality*, 41(2), pp.373-379.

Weigelhofer, G., 2016. The potential of agricultural headwater streams to retain soluble reactive phosphorus. *Hydrobiologia*, 793(1), pp.149-160.

Weigelhofer, G., Leopold, M., Akbari, E., Farnleitner, A. and Kirschner, A., 2022. Was bringen Ufergehölze für die Wasser- und Sedimentqualität von Bächen in landwirtschaftlichen Regionen?. *Österreichische Wasser- und Abfallwirtschaft*.

Wilson, H. F. and Xenopoulos, M. A. 2009. Effects of agricultural land use on the composition of fluvial dissolved organic matter. *Nature Geoscience*, *2*(1), 37-41.

Withers, P. and Jarvie, H., 2008. Delivery and cycling of phosphorus in rivers: A review. *Science of The Total Environment*, 400(1-3), pp.379-395.

Woodward, G., Gessner, M., Giller, P., Gulis, V., Hladyz, S., Lecerf, A., Malmqvist, B., McKie, B., Tiegs, S., Cariss, H., Dobson, M., Elosegi, A., Ferreira, V., Graca, M., Fleituch, T., Lacoursiere, J., Nistorescu, M., Pozo, J., Risnoveanu, G., Schindler, M., Vadineanu, A., Vought, L. and Chauvet, E., 2012. Continental-Scale Effects of Nutrient Pollution on Stream Ecosystem Functioning. *Science*, 336(6087), pp.1438-1440.