DIPLOMARBEIT

„Impact of drying and re-flooding of sediment on the phosphorus dynamics of river-floodplain systems“

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1. Abstract

In the present study we investigated the phosphorus release behaviour of sediments from shallow backwaters of an isolated floodplain of the Danube River situated east of the city of Vienna. We focused on phosphorus dynamics as phosphorus is representing the main limiting nutrient in freshwater systems controlling primary production in freshwater systems. Increased phosphorus concentrations favour algal blooms leading to oxygen depletion and in the case of cyanobacteria potentially to the release of substances toxic to various organisms.

In the light of restoration plans with the aim to initiate erosional processes in this floodplain area by re-establishing hydrological dynamics, the response of sediments to frequent alternations between drying and flooding periods is a key issue as it is known to alter pH, redox conditions, physical properties of minerals and composition and activity of the microbial community. Therefore, these changes are expected to affect the potential for phosphorus release.

In order to determine the effect of changing hydrological conditions on internal phosphorus loading, we exposed sediments to different dry/wet treatments in a laboratory experiment. The duration of the drying period previous to flooding was shown to enhance Total Phosphorus (TP) release from sediments into the water column. Partial Correlation Analysis considering the duration of drying and the sampling site as covariates showed strong and highly significant positive correlations between ΔTP and ΔNH_4^+ as well as between ΔTP and ΔFe. This outcome is indicating that enhanced mineralization rates and the reduction of iron hydroxides leading to a concomitant release of iron and phosphorus are mechanisms responsible for the rise in TP with increasing duration of dry periods.

Alternating drying and flooding periods at intervals of 100 h each over a period of 600 h, leading to a decrease in gravimetric water content from about 50% to 30% during desiccation, had a significant positive effect on TP release upon flooding. Repetition of 200 h drying and 100 h flooding over 600 h, leading to a decrease in gravimetric water content from about 50% to 10% during desiccation, also had a significant positive effect on TP release. This outcome indicates that enhanced hydrological dynamics approaching

the degree of riverine water level fluctuations may initially result in elevated phosphorus concentrations in the floodplain that in consequence might stimulate primary production. Considering longer time periods, this effect is expected to be compensated as the restored floodplain undergoes a regime shift from an isolated system dominated by internal nutrient cycling and local autochthonous production to a connected system controlled by riverine inputs and hydrological dynamics, i.e. shorter retention times and less potential of increased phytoplankton biomass accumulation in the floodplain water bodies.

2. Introduction

Atmospheric deposition and weathering of minerals in the catchment are natural sources of phosphorus in aquatic ecosystems (Reddy & DeLaune, 2008; Carpenter, 2005). Anthropogenic inputs of phosphorus include point source emissions from industrial and municipal waste water treatment plants and diffuse inflow from agriculturally used areas. Since the 1950s nutrient loading has been a major threat to the stability of freshwater ecosystems worldwide, with an expected tendency to gain further weight in the future (Millennium Ecosystem Assessment, 2005). Emissions of phosphorus into the Danube river system rose from 45 kty^{-1} in 1955 to 115 kty^{-1} in 1990 due to an increase in the use of detergents and an expansion of sewerage. From 1990 to 2000 phosphorus emissions dropped down to 68 kty^{-1} due to technical progress in waste water treatment and the replacement of phosphorus in detergents (Kroiss et al., 2005). According to Zessner & van Gils (2002) the mean annual phosphorus load of the Austrian Danube accounts to 7-9 kty^{-1}.

Phosphorus introduced into aquatic systems is either subject to short-term storage mediated by plants and algae or to long-term storage mediated by sediment deposition (Reddy & DeLaune, 2008; Golterman, 2004). Within the Danube river basin 53% of phosphorus emitted to the water are retained in small water bodies along downward transport to the black sea (Kroiss et al., 2005). The main fraction is transported during high flows as particle-bound phosphorus. River floodplain systems play a major role in nutrient retention, especially during high flows as shown for the Dutch Rhine by Van der Lee (2004). The chemical and biotic composition of floodplain water bodies is fluctuating with water levels and therefore largely dependent on connectivity to the main river (Tockner et al., 1999). Periodical changes between inundation and low flow events – referred to as flood pulse by Junk et al. (1989), or as flow pulse as the pulsing of riverine discharge below bankful as flow pulse (sensu Tockner et al. 2000) – are the main characteristics of river floodplain ecosystems. According to the flood pulse concept, the trophic status of connected floodplain waters is mainly influenced by the exchange of dissolved and suspended materials with the main river (Junk et al., 1989). For the availability of phosphorus in shallow water bodies less influenced by riverine dynamics, the process of internal loading from the sediments (i.e. vertical exchange across the sediment-water interface) plays a major role (Søndergaard et al., 2003).

The frequency and pattern of connectivity between the main river and the water bodies of the floodplain determine if external or internal processes control phosphorus dynamics. Lair et al. (2009) reported that sediments of floodplain areas with high connectivity levels to the main channel contain large amounts of inorganic phosphorus due to allochthonous inputs, whereas sediments of areas with lower connectivity contain large amounts of organic phosphorus due to local autochthonous production. Hein et al. (2005) observed a significant positive relationship between discharge level and total phosphorus retention in a reconnected side-arm of the River Danube. In the floodplains of the Atlantic Coastal Plain (USA) phosphorus accumulation rates were positively correlated with the accumulation rates of mineral sediment (Noe & Hupp, 2005). Floodplain sites disconnected from the main river displayed lower accumulation rates regarding sediment and nutrients. Thus, floodplains cut off from the hydrological influence of the main river are no longer able to act as systems of quantitative importance for nutrient retention favouring the water quality of the adjacent water bodies. The implementation of restoration projects with the aim to increase hydrologic connectivity could lead to a regain of this function (Naiman et al.et al., 2005). The effect of flooding events on phosphorus concentrations in the floodplain depends on the duration and magnitude of the flood, as well as on the chemistry of the main river and its floodplain. In the floodplain of the Tualatin River (USA) a high magnitude flood lasting for three months resulted in decreasing total phosphorus concentrations due to a mixing up with nutrient-poor river water and a reduction in algal biomass due to flushing and increasing turbidity. In contrast, a low magnitude flood lasting for one month neither resulted in changing nutrient levels, nor in a shift in the algal community (Weilhoefer et al., 2008). Hein et al. (2005) introduced the parameter "water age" as an inverse measure of residence time describing the temporal development in reconnected side-arms of the River Danube: At water ages above one day the concentration of soluble reactive phosphorus and particulate inorganic matter rapidly decreased, whereas phytoplankton biomass reached its highest densities at water ages between 3 and 10 days (Hein et. al., 2004). The rapid decrease in phosphate concentration with rising water age was attributed to algal uptake (Hein et al., 2004; Naiman et al., 2005).

During flood events riverine suspended or dissolved phosphorus is introduced into the former isolated system directly altering equilibrium conditions for phosphorus exchange within the floodplain. Phosphorus release at the sediment-water interface (“internal loading”) in the floodplain is influenced by lateral exchange processes between the floodplain and the main river. Inflow of water from the main river implicates a change in water chemistry in previously isolated standing waters. This can affect chemical as well as biotic processes controlling internal phosphorus loading from autochthonous sediments. In the backwaters of the Upper Lobau, which are already subject to a controlled water enhancement scheme, lateral water exchange resulted in a well-mixed and oxygenated water column in former isolated backwaters inhibiting phosphorus remobilisation from (anoxic) sediments (Bondar-Kunze et al., 2009).

While the main channel and deeper parts of floodplain lakes remain permanently inundated, riparian zones and shallow areas are periodically falling dry and re-wetting depending on their topography and the surface flow paths. Water level fluctuations and desiccation are expected to have a strong effect on internal phosphorus loading by altering redox conditions of sediments, pH, physical properties of minerals (Lijklema, 1980e; Baldwin, 1996b; Baldwin & Mitchell, 2000b; Darke et al., 1996; Sah et al., 1989) and the composition and activity of the microbial community (Batzer & Sharitz, 2006).

Adsorption and desorption of phosphorus from minerals is redox- and pH-sensitive: For instance, the phosphate binding capacity mediated by Fe minerals decreases with increasing pH value in a pH range between 5 and 8 due to competition of hydroxyl ions for binding sites (Ahlgren, 2006; Lijklema, 1980; Reddy & DeLaune, 2008). At redox potentials below 300 mV or oxygen concentrations below 0.1 mg l^{-1} ferric hydroxides are reduced to ferrous iron by bacteria leading to a release of phosphate that was occluded in the hydrated coatings of the ferric hydroxide (Golterman, 2004; K. Reddy & DeLaune, 2008). During desiccation the altered pore water chemistry and intensified degradation processes result in an accumulation of organic matter leading to rising concentrations of phosphate in the pore water (Golterman, 2004). During exposure to air, ferrous sulphides present in the sediment are oxidized to amorphous ferric (oxy)hydroxides having a large surface area and a very high affinity for phosphorus. However, after prolonged periods of drought, the ferric (oxy)hydroxides become more crystalline resulting in a decrease of binding sites for phosphorus (Lijklema, 1980; Baldwin, 1996).

Desiccation does not only affect sediment and pore water chemistry but is also known to have a strong effect on the microbial community: Bacterial activity decreases linearly with decreasing water content (West et al., 1992; Orchard et al., 1992) and obligate anaerobic bacteria (e.g. iron and sulphur reducing bacteria) are killed during extended drying or form resting stages (Lynch & Hobbie, 1988; Baldwin & Mitchell, 2000). The loss of obligate anaerobic microbes inhibits the reduction of ferric hydroxides and sulphate. Consequently, phosphate remains adsorbed onto ferric hydroxides. Bacteria play a major role in the breakdown of organic matter and thereby have a great impact on nutrient dynamics of freshwater ecosystems (Batzer & Sharitz, 2006).

In summary, the effects of drying and re-flooding of sediments on phosphorus dynamics can be ascribed to changes in the bacterial community as well as to changes in water chemistry. Although there is much evidence that hydrological changes have a strong impact on the process of internal loading of phosphorus from floodplain sediments, no laboratory experiment evaluating this impact by simulating alternating periods of wettingdrying and has so far been conducted for different floodplain sediment types.

Therefore we conducted a laboratory experiment addressing the following questions:

(i) Do drying periods lasting for different time periods previous to flooding affect the phosphorus release behaviour of sediments?

(ii) Does repeated drying and wetting have an impact on the phosphorus release behaviour of sediments?

Thus, the objective of this study was to assess the impact of changing hydrological conditions on internal phosphorus loading by sediments of isolated backwaters. We expected phosphorus release to increase with increasing periods of drought as well as with repeated drying and wetting periods. In our laboratory experiment the effect of lateral exchange between the floodplain and the main river on vertical exchange ("internal loading") was considered by using water from the river Danube for wetting of sediment cores taken from backwaters. The objective of this study was to assess the impact of changing hydrological conditions on internal phosphorus loading by sediments of isolated backwaters.

3. Material and Methods

3.1 Study site

The floodplain of the Lower Lobau (48°10’N 016°30’E), stretching an area of 915 ha, was disconnected from the main channel of the River Danube by flood protection measures by the end of the nineteenth century. Today, water supply is mainly mediated by groundwater and seepage inputs as the “Schönauer Schlitz” (interruption of dam, see Figure 1) represents the only residual site providing surface water connection to the main river. Sedimentology of the water bodies of the Lower Lobau, which are isolated from each other by cross bars or fords, is determined by their distance from the “Schönauer Schlitz” reflecting hydrological connectivity: The thickness of the fine sediment layer decreases and the organic content of the sediment increases with increasing distance from the “Schönauer Schlitz” (Reckendorfer & Hein, 2000).

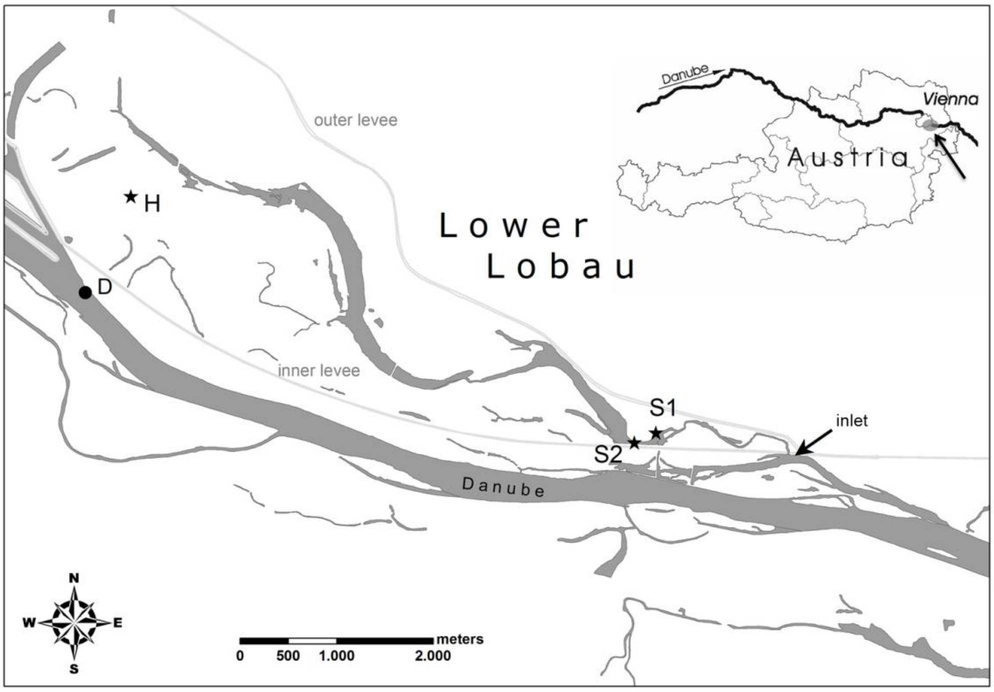

Figure 1 Study area Lower Lobau and the sampling sites: “Hanslgrund“ (H), “Schönauer Wasser” (S1, S2), water for wetting (D). Surface water connection between the Danube River and the sampling sites is only possible when water flows back through the inlet (“Schönauer Schlitz”) interrupting the flood protection levee.

Although altered in many respects, the Lower Lobau still exhibited a high nature value and was thus declared a UNESCO biosphere reserve in 1977, designated a “Wetland of International Importance” after the Ramsar Convention in 1982, and finally became part of the National Park Donau-Auen founded in 1996. Besides its importance as a nature conservation area, the Lower Lobau is subject to small-scale fishing and forestry, serves as drinking water source for the city of Vienna during periods of high supply and is used for recreation (Hein et al. 2006). In order to reduce siltation processes threatening the valuable water bodies of the Lower Lobau, plans to enhance surface water connectivity with the River Danube and to improve integration between currently separated basins have been developed and implemented in parts of the area (*Managementplan Nationalpark Donau-Auen 2009-2018*, 2009). The implementation of a controlled surface water exchange programme for the Lower Lobau – like already realized for the Upper Lobau – with the aim to restore dynamic alluvial processes, to maintain habitat heterogeneity, to protect endangered species and also to maintain ecosystem services like drinking water supply and recreation (Baart et al., 2010; Hein et al., 2006) is currently being tested.

For the collection of sediment samples to be used in our experiment, water bodies differing in surface water connection to the River Danube and in organic content of sediments were chosen as these characteristics were assumed to be major determinants for phosphor remobilisation. Sediment samples were taken from two different shallow backwaters called “Schönauer Wasser“ and ”Hanslgrund“ (Figure 1, Table 1). Surface water connections to the River Danube are prevailing on 137 da^{-1} at “Schönauer Wasser“ and on 4 da^{-1} at “Hanslgrund“. Within “Schönauer Wasser“, two sites differing in organic content (“S1“, “S2“, Figure 1, Table 3) were chosen. All sampling sites are exposed to drought for several weeks during summer at low flow.

Table 1 Chemical characteristics of the study areas according to Reckendorfer & Hein (2000) and unpublished monitoring data from 2006-2009 (median ± interquartile range, n=45).

	Schönauer Wasser	Hanslgrund
Orthophosphate (PO_4^{3-})	1.2 ± 1.62 µg l^{-1}	28.4 ± 49.4 µg l^{-1}
Total Phosphorus (TP)	45.0± 23.6 µg l^{-1}	61.1 ± 93.6 µg l^{-1}
Ammonium (NH_4^+)	24.3 ± 20.4 µg l^{-1}	27.7 ± 51.1 µg l^{-1}
Nitrite (NO_2^-)	3.0 ± 7.0 µg l^{-1}	1.2 ± 5.2 µg l^{-1}
Nitrate (NO_3^-)	102.8 ± 323.3 µg l^{-1}	102.8 ± 57.1 µg l^{-1}
pH	8 ± 0.4	7.6 ± 0.3

3.2 Field sampling

Sampling of sediment cores was carried out in July 2010. Per sampling site, fifteen sediment cores with a diameter of 15 cm, a length of 10 cm and a fresh weight of approximately 900 g were taken using a PVC tube. According to Boström et al. (1982), the upper 10 cm of the sediment layer play a role in the nutrient balance of lake ecosystems. The sediment cores were transferred into plastic vessels without disturbing the surface layer. During the two-hour transport to the laboratory the sediment samples were kept in the dark and at constant temperature. Water from the River Danube (Figure 1) was collected 30 cm below surface.

3.3 Analyses of sediment samples

From each of the 45 sediment cores, subsamples for analysis of organic content, amorphous iron and phosphorus fractions were taken immediately after field sampling and kept deep-frozen for four weeks. Organic content was determined by the loss on ignition method (450°C, 4h). Amorphous iron was extracted by adding 30 ml of a solution containing 16.2 g of $(COONH_4)_2.H_2O$ and 10.9 g of $(COOH_2).2\ H_2O\ l^{-1}$ to 2 to 2.5 g of sediment (Loeb et al., 2008; Schwertmann, 1964). The iron concentration was determined according to US Standard Methods 3500-Fe D and DIN 38406-E1-1 using a Hitachi U-2000 photometer. Inorganic phosphorus, organic phosphorus, soluble reactive phosphorus, Fe- and Mn-bound phosphorus, Al-bound phosphorus and Ca- and Mg-bound phosphorus were determined according to the protocol of (Ruban et al. (2001). Each fraction was extracted separately and at least 3 g of dry sediment for each extraction were used. Sediment with reagent was shaken for 16 h and centrifuged at 3000 rpm for 15 min at 20°C. Total phosphorus was extracted by adding 10 ml of HNO_3 (65%) to 0.3 g to 0.4 g of sediment and putting it into a microwave (CEM MarsXpress, 10 min heating-up to 175°C). Afterwards, phosphorus concentrations were determined by Continuous Flow Analysis (CFA) by a Systea 3rd Generation Continuous Flow Analyser (Alliance instruments) according to ÖNORM EN ISO 15681-2.

3.4 Experimental setup

During the experiment, the sediment cores were kept in two-litre plastic vessels and incubated in the dark at a temperature of 30°C in the dry box. Three sediment cores per sampling site were randomly assigned to one out of four dry/wet cycles (Figure 2).

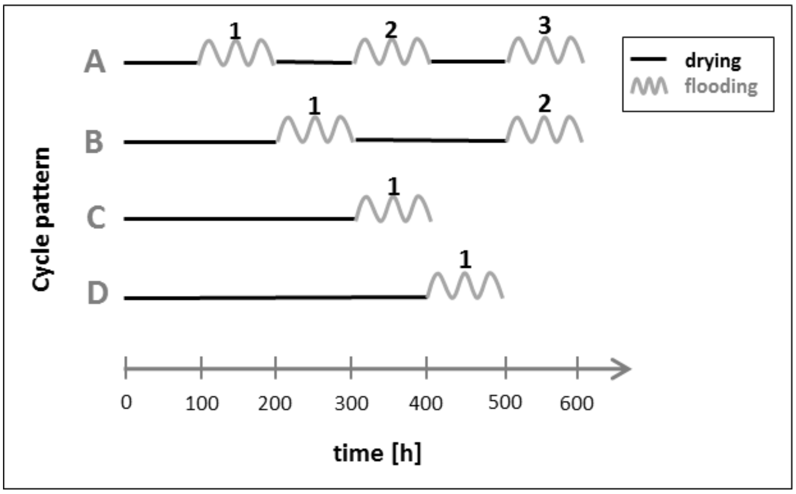

Figure 2 Scheme displaying four different dry/wet cycle patterns (A,B,C,D) simulated in the dry box at a constant temperature of 30°C. Wetting periods are consecutively numbered within each dry/wet cycle pattern.

The first wetting periods of dry/wet cycles A, B, C and D were simulated after 100 h, 200 h, 300 h and 400 h of drying. This setup was chosen in order to evaluate the effect of different periods of drought previous to flooding on phosphorus release. In order to find out if repeated drying and wetting has an impact on phosphorus release, multiple flooding periods were simulated within dry/wet cycles A and B: Dry wet/cycle A was composed of three drying periods of 100 h each, alternating with three flooding periods of 100 h each. Dry wet/cycle B was composed of two drying periods of 200 h each, alternating with two flooding periods of 100 h each.

The duration of flooding periods was set to 100 h as in experiments carried out by Turner & Haygarth (2001) maximal phosphorus release attributed to lysis of bacterial cells in soils dried at 30°C for seven days occurred within the first three days of re-wetting. Furthermore, according to Reddy & DeLaune (2008), reduction of iron and concomitant phosphate release starts after the reduction of the entire nitrate pool which was observed to be completed three days after inundation in soils of the Lake Apopka marsh. We assume this process to be similar in floodplain ecosystems as a negative correlation between the concentration of nitrate and the point of time at which phosphorus release started was observed for river riparian sediments by Surridge et al. (2007). For wetting, water from the River Danube was used, since this is the water entering the investigated water bodies during flood events. The water was filtered through double Whatman GF/F filters and stored at 4°C. Two days before its addition to the sediment cores, the water was treated with anion exchange resins (DOWEX Marathon A; 200 g per 20 litres of water) in order to reduce the concentration of PO_4^{3-} and to thereby render the effect of phosphorus release from the sediment more apparent. Sediment cores were filled with one litre of treated river water.

Table 2 Nutrient concentration of Danube River water treated with anion exchange resins before being used in the experiments (n=5).

Parameter	Mean	Standard deviation
TP [µg l^{-1}]	7.8	± 4.3
PO_4^{3-} [µg l^{1}]	7.0	± 5.5
NO_3^- [µg l^{-1}]	881.0	± 175.9
NO_2^- [µg l^{-1}]	4.7	± 0.3
NH_4^+ [µg l^{-1}]	14.0	± 4.4
Fe [mg l^{-1}]	< 0.2 (below detection limit)	-
pH	9.44	± 0.43

Wetting periods were terminated by withdrawing the water with a peristaltic pump. Before and after 100 h of flooding, surface water samples for the determination of TP, PO_4^{3-}, NO_3^-, NO_2^- and NH_4^+, and $Fe^{2+/3+}$ were taken and filtered through Whatman GF/F filters. Unfiltered samples were used for Total Phosphorus (TP) determination.

3.5 Analyses of water samples

Nutrient concentrations were determined according to ÖNORM EN ISO 15681-2 (PO_4^{3-}), DIN EN ISO 13395 (NO_3^- and NO_2^-) and DIN EN ISO 11732 (NH_4^+) by Continuous Flow Analysis using a Systea Alliance instrument

Total Phosphorus was digested with persulfate according to Eaton & Franson (2005) and subsequently analysed following the same procedure as for PO_4^{3-}. Concentrations of $Fe^{2+/3+}$ were determined with a Lange LCK 320 cuvette test using a Hach DR-2800 portable photometer.

3.6 Statistical Analysis

Statistical analysis was conducted using SPSS 16.0 for Windows. For the analysis the differences between the concentrations before and 100 h after flooding regarding TP, PO_4^{3-}, NO_3^-, NO_2^-, NH_4^+, and Fe were used.

Kruskal Wallis Tests were performed in order to find out if sampling sites differed regarding their organic content and phosphorus fractions. When the Kruskal Wallis Test showed significant results, a Mann Whitney Test was carried out for pairwise comparisons of the three sampling sites.

For statistical evaluation of the impact of drying previous to flooding, data from the first flooding periods of dry/wet cycle patterns A, B, C and D (Figure 2) were analysed. First, a Kruskal Wallis Test was performed in order to determine if the dry treatment previous to flooding and/or the sampling site affected the release or uptake of TP, PO_4^{3-}, Fe, NH_4^+, NO_3^- and NO_2^- upon flooding. As the factor "sampling site" did not have a significant effect on the change in TP, NH_4^+ and Fe concentrations, further statistical analysis was conducted without splitting data according to sampling sites. A Mann Whitney Test was carried out for pairwise comparison of dry treatments. Partial Correlation considering the dry treatment and the sampling site as covariates was performed in order to evaluate relationships between TP and NH_4^+ as well as between TP and Fe. In order to statistically evaluate the effect of repeated drying and wetting, data from multiple wetting periods simulated within dry/wet cycle patterns A and B (Figure 2) were analysed.

For dry/wet cycle A – including three different wetting events – a Friedman Test was performed for comparisons regarding TP, NH_4^+, and Fe. When the Friedman Test resulted in significant differences, a Wilcoxon Test for pairwise comparisons was conducted in order to evaluate which of the three flooding events differed from each other.

For dry/wet cycle B – including two different wetting periods – a Wilcoxon Test was performed for testing differences regarding TP, NH_4^+ and Fe.

4. Results

4.1 Sediment characteristics

Sediment samples of the three sites did not differ regarding Total Phosphorus (Kruskal Wallis, n=45, χ^2=3.366, df=2, p=0.186), Organic Phosphorus (Kruskal Wallis, n=45, χ^2=3.913, df=2, p=0.141) and Inorganic Phosphorus (Kruskal Wallis, n=45, χ^2=1.157, df=2, p=0.561).

Inorganic Phosphorus accounted for 60% of Total Phosphorus in "Schönauer Wasser 1" and "Schönauer Wasser 2" and for 50% in "Hanslgrund"(Table 3).

The three sampling sites significantly differed from each other in organic content (Table 4).

Sampling site "Schönauer Wasser 2" significantly differed from the sampling sites "Schönauer Wasser 1" and "Hanslgrund" in Mn- and Fe-bound phosphorus (Table 4).

Sampling site "Hanslgrund" significantly differed from the sampling sites "Schönauer Wasser 1" and "Schönauer Wasser 2" in Amorphic Iron content, Soluble Reactive Phosphorus, and Ca- and Mg-bound P (Table 4).

Table 3 Mean **s**ediment composition of the three sampling sites regarding organic content, amorphic iron, and different phosphorus fractions (mean ± standard deviation, n=15).

	S1	S2	H
Organic content [%]	10.1 ±3.6	7.0 ± 0.8	19.0 ± 5.7
Amorphic Iron [g kg^{-1}]	3.97 ± 0.93	3.60 ± 0.41	1.66 ± 0.27
Total P [mg kg^{-1}]	543.1 ± 94.9	548.3 ± 31.6	580.8 ± 70.3
Organic P [mg kg^{-1}]	159.1 ± 33.8	155.4 ± 49.5	196.5 ± 71.3
Inorganic P [mg kg^{-1}]	391.5 ± 18.0	385.6 ± 17.9	389.3 ± 19.3
SRP [mg kg^{-1}]	0.8 ± 0.3	0.9 ± 0.3	2.7 ± 1.6
Mn- and Fe- bound P [mg kg^{-1}]	50.7 ± 19.9	74.7 ± 13.0	50.8 ± 17.0
Ca- and Mg- bound P [mg kg^{-1}]	354.3 ± 23.8	343.7 ± 17.7	371.2 ± 20.0

Table 4 Results of Mann-Whitney Test, pairwisely comparing sediment composition of the three sampling sites (n=15). Total P, Organic P, and Inorganic P are not included as the Kruskal Wallis Test did not show any significant differences among sampling sites for these phosphorus fractions.

Pairwise comparison of	Statistical parameters	Organic content	Amorphic Iron	SRP	Mn- and Fe-bound P	Ca- and Mg-bound P
S1-S2	*U*	40.5	90	84	41	73
	p	0.003	0.351	0.237	0.003	0.101
S1-H	*U*	28	0	25	109	64
	p	0.0005	0.000003	0.0003	0.885	0.044
S2-H	*U*	21.5	0	28	31	34
	p	0.0002	0.000003	0.0005	0.001	0.001

4.2 Impact of drying periods

The impact of drying on nutrient release upon re-flooding was evaluated by comparing wetting periods A1, B1, C1, and D1 (Figure 2) that occurred after 100 h, 200 h, 300 h, and 400 h of drought, respectively. The concentrations of TP, Fe, NH_4^+, NO_3^-, and NO_2^- in the water column were highly significantly influenced by the duration of the drying period previous to flooding (Table 5), whereas the sampling site did not have any significant effect (Table 6). In consequence, further analysis regarding TP, Fe, NH_4^+, NO_3^-, and NO_2^- was conducted pooling together data from all sampling sites.

In contrast to other nutrients measured in the water column, the concentration of PO_4^{3-} significantly differed among sampling sites (Table 6), but not among dry treatments (Table 5). Due to this outcome and as this work focuses on the effect of dry/wet cycles on nutrient release, $PO_4^{3-}$was not considered in further statistical analysis.

Table 5 Kruskal Wallis Test with “dry treatment“(100 h/ 200 h/ 300 h/ 400 h of drying, see Figure 2) as group variable (n=36).

	ΔTP	ΔPO_4^{3-}	ΔFe	ΔNH_4^+	ΔNO_3^-	ΔNO_2^-
χ^2	27.789	4.216	26.669	29.222	30.611	11.537
df	3	3	3	3	3	3
p	0.00004	0.239	0.000007	0.000002	0.000001	0.009

Table 6 Kruskal Wallis Test with “sampling site“ (Schönauer Wasser 1, Schönauer Wasser 2, Hanslgrund, see Figure 1) as group variable (n=36).

	ΔTP	ΔPO_4^{3-}	ΔFe	ΔNH_4^+	ΔNO_3^-	ΔNO_2^-
χ^2	2.252	13.781	1.880	0.221	0.011	2.254
df	2	2	2	2	2	2
p	0.324	0.0001	0.391	0.896	0.994	0.324

The longer the previous drying period lasted, the higher the concentrations of TP, Fe, and NH_4^+ were during re-wetting (Figure 3). Significant differences between each of the drying treatments were observed regarding TP, Fe, and NH_4^+ (Table 7). Iron concentrations of all replicates dried for 100 h and 200 h previous to wetting and of two replicates dried for 300 h were below the detection limit of the method used.

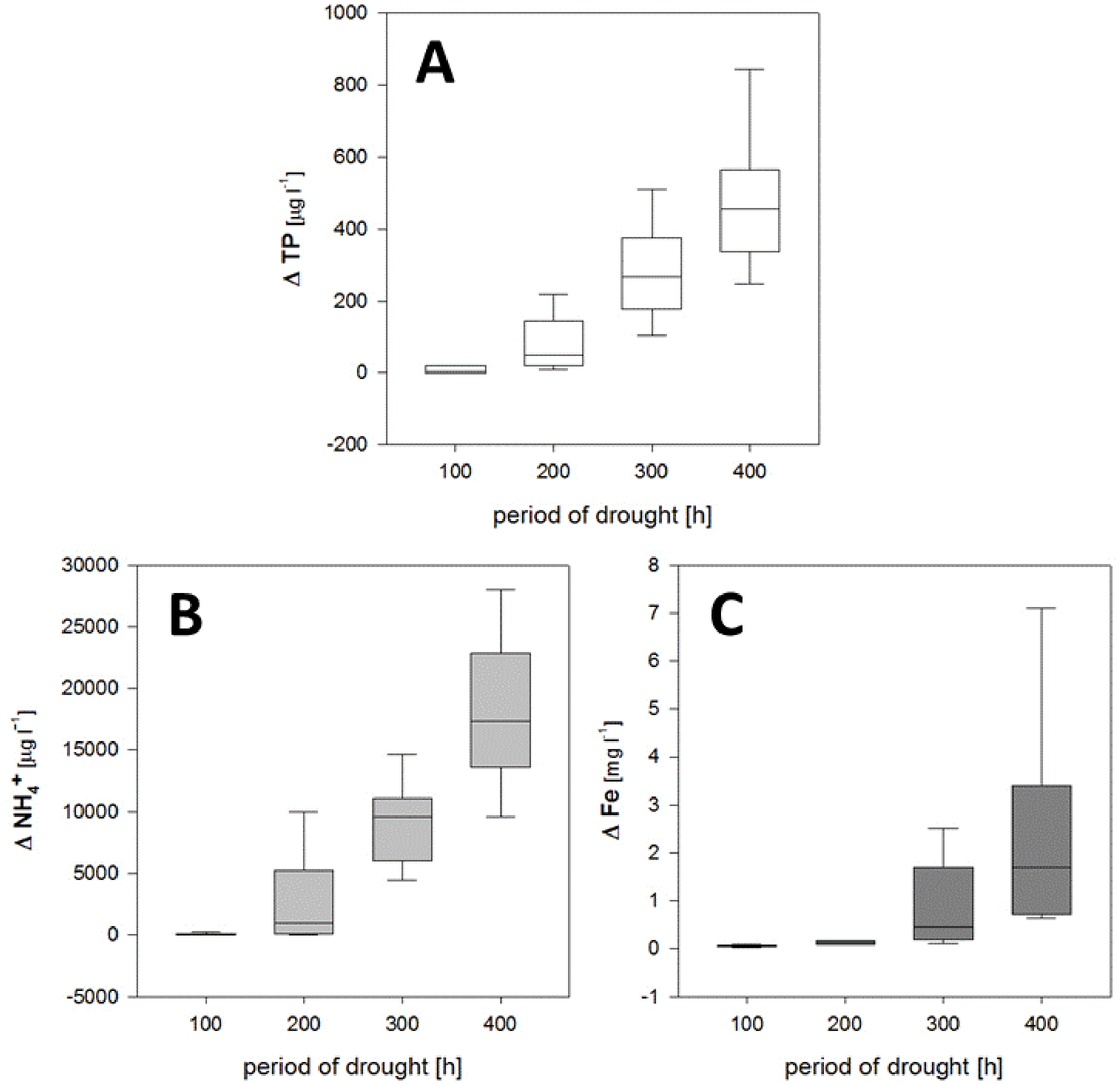

Figure 3 Boxplots displaying ΔTP (A), ΔNH_4^+ (B), and Δ Fe (C) after 100 h of wetting for four different drying treatments: A1 – 100 h, B1 – 200 h, C1 – 300 h, D1 – 400 h of drying at 30°C previous to wetting (n=9). Δ refers to the difference between the concentration at the start and at the end of the wetting period. Data of all sampling sites were pooled (n=9).

Table 7 Mann Whitney Test for pairwise comparison of drying treatments (A1 = 100 h, B1 = 200 h , C1 = 300 h, D1 = 400 h of drying at 30° previous to wetting). Data of all sampling sites were pooled (n=9).

		A1-B1	A1-C1	A1-D1	B1-C1	B1-D1	C1-D1
ΔTP	*Whitney U*	7	0	0	6	0	13
	p	0.005	0.001	0.001	0.002	0.004	0.015
ΔFe	*Whitney U*	8	0	0	9	0	17
	p	0.007	0.0004	0.0004	0.009	0.001	0.038
ΔNH_4^+	*Whitney U*	8	0	0	7	39	6
	p	0.005	0.0004	0.0004	0.003	0.0000001	0.002

Partial Correlation Analysis considering the dry treatment and the sampling site as covariates in order to avoid spurious correlation showed strong and highly significant positive correlations between ΔTP and ΔNH_4^+ (r=0.744, df=31, p=0.000001) as well as between ΔTP and ΔFe (r=0.441, df=31, p=0.01) (Figure 4).

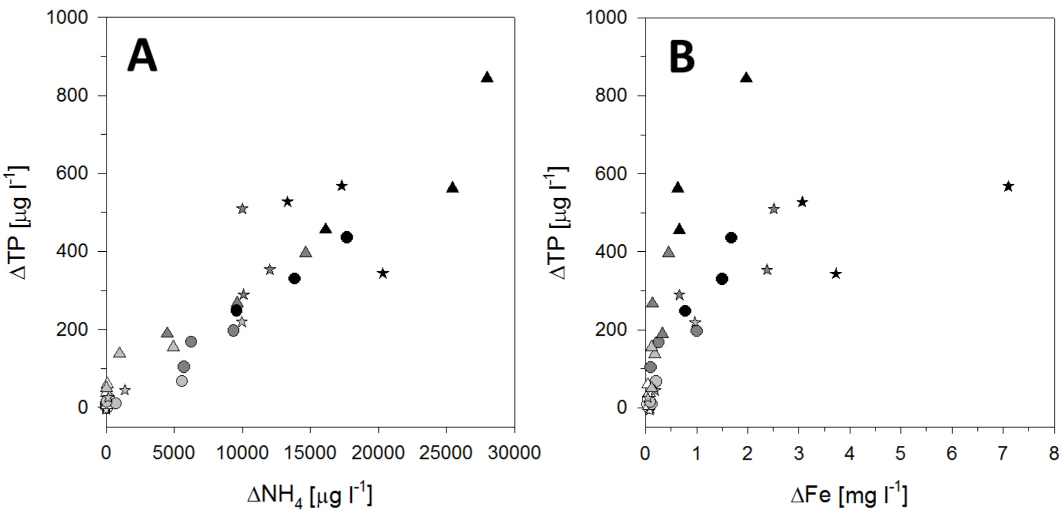

Figure 4 Scatterplots showing the relationship between Δ TP and Δ NH_4^+ (A) and the relationship between ΔTP and ΔFe (B) upon flooding. Three different sampling sites (○ = Schönauer Wasser 1, ☆ = Schönauer Wasser 2, Δ = Hanslgrund) and four different dry treatments (white = 100 h, grey = 200 h, dark grey = 300 h, black = 400 h of drought at 30°C previous to flooding) are included. Highly significant strong positive correlations were observed for (A) r=0.744, df=31, p=0.000001 and (B) r=0.441, df=31, p=0.01 (Partial Spearman Correlation, covariates=dry treatment, sampling site).

4.3 Impact of repeated drying and wetting cycles

Within dry/wet cycle pattern A (Figure 2), including drying periods of 100 h alternating with flooding periods of 100 h, ΔTP differed significantly between the first and the third flooding event (Wilcoxon, n=9, Z=-0.169, p=0.018) as well as between the second and the third flooding event (Wilcoxon, n=9, Z=-0.169, p=0.018). No significant differences regarding ΔTP were observed between the first and the second flooding event (Wilcoxon, n=9, Z=-0.169, p=0.866). ΔTP showed a five-fold increase in the third flooding event (mean = 57.20 µg l^{-1}) compared to the two preceding ones (means = 11.59 µg l^{-1}, 10.58 µg l^{-1}, respectively). ΔNH_4^+ did not differ significantly among flooding events of dry/wet cycle A (Friedman, n=9, χ^2=0.750, df=2, p=0.687). The rise in the concentration of Fe was significantly lower in the first compared to the third flooding event (Wilcoxon, n=9, Z=-2.666, p=0.008) and in the second compared to the third flooding event of dry/wet cycle A (Wilcoxon, n=9, Z=-2.547, p=0.011). The change in the concentration of Fe showed a two-fold increase in the third flooding event (mean = 0.151 mg l^{-1}) compared to the two preceding ones (means=0.06 mg l^{-1}, 0.07 mg l^{-1}, respectively). It has to be remarked, that Fe concentrations of all replicates in this treatment group went below the detection limit of the method used.

Within dry/wet cycle pattern B (Figure 2), including drying periods of 200 h alternating with wetting periods of 100 h, flooding periods significantly differed regarding ΔTP (Wilcoxon, n=9, Z=-2.192, p=0.028) as well as ΔNH_4^+ (Wilcoxon, n=9, Z=-2.192, p=0.028). Means of both variables showed a four-fold increase in the second flooding event (ΔTP = 352. 09 µg l^{-1}, ΔNH_4^+= 10022.46 µg l^{-1}) compared to the first one (ΔTP = 80.01 µg l^{-1}, ΔNH_4^+= 2640 µg l^{-1}). No significant differences were observed regarding ΔFe among flooding events of dry/wet cycle B (Wilcoxon, n=9, Z=-1.244, p=0.214). Iron concentrations of all replicates in the first wetting period and of one third of the replicates in the second wetting period went below the detection limit of the method used.

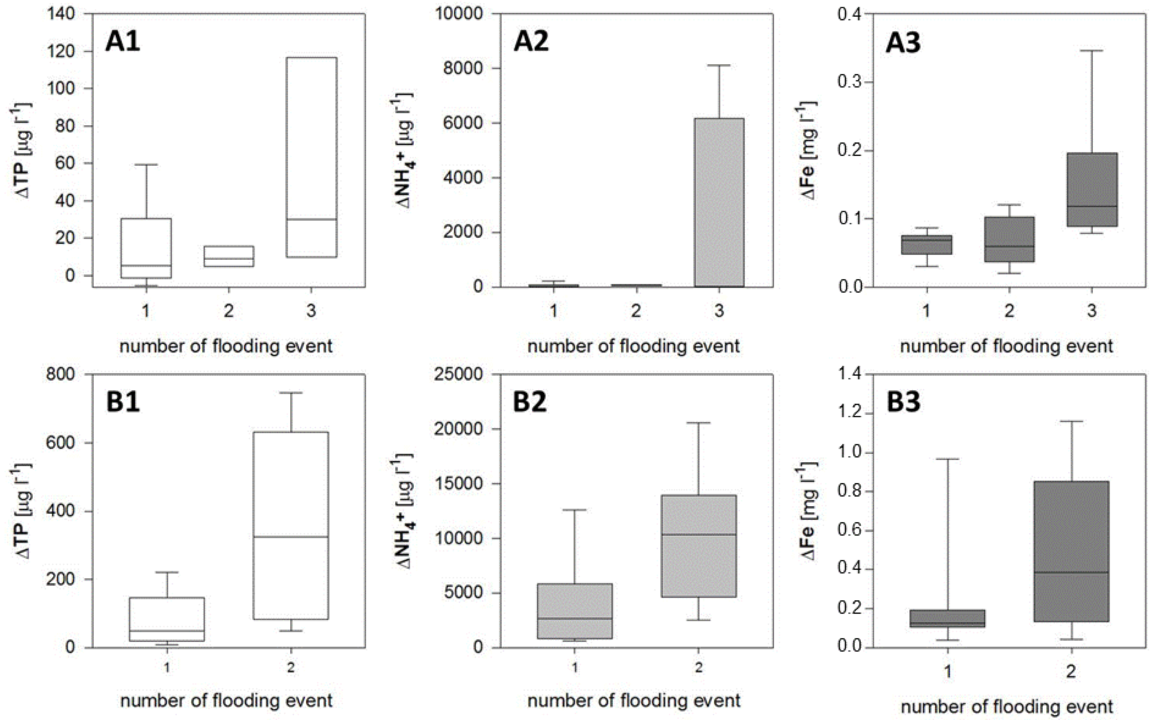

Figure 5 Boxplots showing ΔTP (1), Δ NH_4 (2), and ΔFe (3) after 100 h of wetting for different wetting events of dry/wet cycle A (A) and B (B). Data of all sampling sites were pooled (n=9).

Table 8 pH and O_2 concentration after 24 h and 100 h of wetting expressed as mean±standard deviation (n=9). Values shown represent means of all sampling sites pooled.

Cycle	Flooding nr.	pH	O_2 [mg l^{-1}]
A	**1**	-	-
A	**2**	8.08±0.08	5.03±0.54
A	**3**	8.03±0.09	4.42±1.25
B	**1**	7.93±0.05	3.97±1.04
B	**2**	7.97±0.08	2.54±1.56
C	**1**	7.93±0.08	2.12±0.95
D	**1**	7.84±0.08	1.01±0.62

5. Discussion

5.1 Impact of drying periods

The impact of drying on nutrient release was evaluated by comparing concentrations of TP, NH_4^+ and Fe during wetting phases that were simulated after different periods of drought (A1 – 100 h,.B1 – 200 h, C1 – 300 h, D1 – 400 h, Figure 2). In our experiments, release of TP, NH_4^+, and Fe to the water column during wetting phases increased with increasing duration of the preceding drying periods (Figure 3).

5.1.1 Phosphorus release mediated by mineralization of organic matter

The relationship between NH_4^+ and TP was characterised by a strong and highly significant correlation (Figure 3), indicating that mineralization processes could be an important mechanism responsible for the rise in TP release. Regarding this issue, one question remains: Why did drying not lead to elevated PO_4^{3-} concentrations in the water column? One explanation could be that mineralization to PO_4^{3-} took place, but could not be detected as PO_4^{3-} was subject to fast uptake by microbes and benthic algae and therefore rapidly became part of the organic phosphorus pool. In microcosms with decomposing plant material, Barsdate et al. (1974) observed uptake velocities of 1.38-16.6 x 10^{-7} µg PO_4^{3-} h^{-1} $bacteria^{-1}$ at PO_4^{3-} concentrations of more than 64 µg l^{-1}. Therefore, PO_4^{3-} may not be measurable due to its rapid transformation into organic phosphorus and may thus only be detectable as part of TP. Another aspect addressing this issue could be that transformation of organic phosphorus compounds into PO_4^{3-} takes longer than ammonification of organic nitrogen compounds. This assumption is supported by the short turnover time observed for proteins (130-310 h), amino acids (87-148 h) and urea (34-53 h) determined in littoral and profundal sediments of Blelham Tarn (English Lake District) by Ahlgren et al. (2005) and Reitzel et al. (2007) as well as by high half-life times of several years to decades for organic phosphorus compounds determined in sediments of mesotrophic Lake Erken (Sweden) by Turner & Haygarth (2001): 21 years for orthophosphate monoesters (e.g. sugar phosphates, mononucleotides, inositiol phosphates), 23 years for orthophosphate diesters (e.g. phospholipids, teichoic acid, desoxyribonucleic acid), and two years for polyphosphates (e.g. ATP).

Regardless of this problem the question remains why mineralization rates increase with increasing periods of drought. In a laboratory experiment, Turner et al. (2003) dried pasture soils for seven days at 30°C and found out that 95% of Molybdate Unreactive Phosphorus (i.e. organic phosphorus and inorganic polyphosphates) released upon re-wetting could be related to the release of phospholipids and nucleic acids due to lysis of bacterial cells during the drying period or due to an osmotic shock upon re-wetting. According to Turner & Haygarth (2001), 56-100% of phosphorus released after flooding of previously dried grassland soils (7 days, 30°C) were part of the organic phosphorus pool. Furthermore, Turner & Haygarth (2001) found a positive correlation between water-soluble organic phosphorus and microbial phosphorus. Qiu & McComb (1995) estimated that air-drying of lake sediment killed 76% of the microbial biomass, governing a five-fold increase in dissolved phosphorus concentration upon flooding. Upon re-wetting, previously dried soils are characterized by a high activity of the newly establishing community of microbes and a high availability of fresh dead organic matter. These conditions are favouring high mineralization rates and in consequence release of ammonium (Birch, 1960). Another cause for ammonium accumulation in sediments upon drying could be the fact that nitrification of ammonium is no longer taking place as nitrifying bacteria are much more susceptible to dehydration than ammonium producing bacteria (Blume et al., 2010). Taking these studies into account, gradual increases in TP and NH_4^+ release with increasing drying periods could be the result of increasing numbers of lysed microbial cells serving as substrate for those members of the microbial community that survived the stress of drying. Decreasing oxygen concentrations with increasing periods of drought (Table 8) might support this hypothesis as oxygen depletion is likely to occur when aerobic heterotrophic organisms show high metabolic activities and therefore have a high oxygen demand. The formation of anaerobic micro-areas in the sediment under low oxygen conditions in the water column could be another factor triggering NH_4^+ release from sediments as oxidation of NH_4^+ to NO_3^- (nitrification) is reduced under anaerobic conditions, resulting in the release of NH_4^+ following decomposition of organic matter (Naiman et al., 2005). Our results suggest that desiccated sediments of former isolated backwaters are acting as source of nutrient release during the first phases of flooding events after the implementation of restoration measures.

5.1.2 Phosphorus release mediated by reduction of iron hydroxides

According to Lijklema (1980), 50% of the phosphate binding capacity of sediments of shallow lakes and reservoirs in the Netherlands is mediated by amorphous iron species, exhibiting a specific surface area of 200-600 m^2 g^{-1} (Blume et al., 2010). Depletion of oxygen in the water column with increasing drying periods coincides with low redox potentials, favouring phosphorus release due to reduction of iron hydroxides. Ferric iron is reduced in contact with bacterial enzymes and not indirectly in the presence of reducing metabolites at low redox potentials (Munch & Ottow, 1983). Amorphous iron(III) hydroxides are the only species of iron accessible to microbial reduction (Lovley & Phillips, 1986; Lovley & Phillips, 1987; Munch & Ottow, 1983). In our experiments release of Fe and TP increased with increasing periods of drought previous to flooding. The relationship between Fe and TP was characterised by a significant positive correlation. Reduction of Fe takes place when redox potentials fall below 300 mV (Golterman, 2004). In the present study we did not measure redox potentials, but decreasing oxygen concentrations in the water column with increasing duration of drought periods co-occurred with elevated iron release.

5.1.3 Phosphorus release due to loss of phosphorus sorption capacity

Drying events lead to an increase in crystallinity of iron hydroxides and therefore result in a loss of the phosphorus binding capacity of sediments (Mc Laughlin et al., 1981). In agreement with this, Darke & Walbridge (2000) showed that iron oxides and hydroxides in high elevation sites of the Ogeechee river floodplain in the United States exhibited higher crystallinity than low elevation sites that are exposed to dry conditions for shorter time periods. Submerged sediments of a water storage reservoir in New South Wales (Australia) exhibited higher ratios of amorphous to crystalline iron and therefore higher phosphate adsorption capacities than wet sediments from the littoral zone and dry sediments from sites exposed to air (Baldwin, 1996). In a laboratory experiment conducted by Qiu & McComb (1994), drying of lake sediment for 40 days at 20°C resulted in elevated phosphorus loading to the water column, which was also related to a loss of sorption capacity of the sediment due to increased crystallinity of iron oxides and hydroxides. In another study Qiu & McComb (2002) exposed sediments of seven shallow lakes in Western Australia to air-drying and observed that 72% of the variation in phosphorus sorption capacity could be explained by the variation of iron crystallinity upon drying. Increasing TP release to the water column with more extended drying in our experiment could therefore be attributed to a gradual increase in crystallinity of iron species.

5.2 Impact of repeated drying and wetting cycles

Within dry/wet cycle pattern A (Figure 2), including drying periods of 100 h alternating with flooding periods of 100 h, TP release was low compared to other treatment groups. The third flooding period showed significantly higher values of ΔTP and ΔFe than the first and the second flooding period. For ΔNH_4^+, no significant differences could be detected, but the trend appears to be similar to that of ΔTP and ΔFe with the highest release rates occurring within the third flooding period. We assume that dry/wet cycle A did not subject the microbial community to severe dry stress as gravimetric water content within 100 h of drying decreased only slightly from about 50% to 30%. Therefore, within dry/wet cycle A, we did not expect large inputs of TP and NH_4^+ due to the reduction of microbial biomass by dehydration. The slight increase in ΔTP in the third flooding period compared to the first and second flooding period could be the result of an elevated number of lysed cells due to the fast alternation of dry and wet conditions. This is supported by a slight increase in ΔNH_4^+ and ΔFe, which can be ascribed to a slightly enhanced mineralization activity and oxygen consumption by the surviving microbial community.

Within the second flooding period of dry/wet cycle pattern B (Figure 2), including drying periods of 200 h alternating with wetting periods of 100 h, ΔTP, ΔNH_4^+, and ΔFe reached levels similar to that of sediments flooded after a drying period of 300 h (C1, Figure 2). We assume that dry/wet cycle B has caused severe drying stress to microbes as gravimetric water content decreased from about 50% to 10% within a drying period of 200 h. In total, microbes were subjected to two drying periods lasting for 200 h each, interrupted by two flooding periods of 100 h each. Within a drying period of 300 h gravimetric water content decreased from 50% to nearly 0%. Therefore, the intensity of drying stress was higher prior to flooding within dry/wet cycle C, but was acting for longer time periods within dry/wet cycle B prior to the second flooding period. In addition to the negative effects of extended drying, microbes had to deal with fast changes between conditions of severe dryness and flooding within dry/wet cycle B. The similarities in pH and O_2 concentrations (Table 8) between flooding periods C1 and B2 further underline the fact that biota were affected in a similar manner by these different treatments.

In summary, repeated drying and wetting resulted in elevated phosphorus release. This effect was more pronounced when drying periods lasted for 200 h than when they lasted 100 h, indicating that the duration of drying is a major determinant controlling phosphorus release upon re-flooding.

The implementation of a surface water exchange program in the study area Lower Lobau is going to result in more frequent water level fluctuations. Riparian areas are going to face faster successions between periods of desiccation and inundation. In the light of the present results, restoration efforts are assumed to lead to enhanced phosphorus loading from sediments during the first phase in the floodplain, which could stimulate primary production and favor increased algal biomass. As sediments of the Lower Lobau are very rich in phosphorus, they are expected to be an important source of this nutrient when the floodplain area is subjected to frequent drying and re-wetting events.

Higher connectivity levels following restoration efforts are known to lead to a shift from an isolated floodplain dominated by internal nutrient cycling (autochthonous production) to a connected floodplain characterised by frequent and intense exchange of nutrients with the main river (Felkl, 2011; Lair et al., 2009). Therefore, considering long-term effects, a reconnection of the Lower Lobau is not expected to trigger eutrophication processes, but to enforce export of developed algal biomass into the river system and thereby preventing accumulation of sediments rich in organic components that are representing a major source of phosphorus release. Furthermore, Baldwin & Mitchell (2000) suggested that repetition of drying and wetting over longer time periods will select for bacterial r-strategists that are reproducing fast when favourable environmental conditions occur and can outlive unfavourable conditions by producing resting stages. Taking our results into account, this could lead to a stagnation of nutrient release as dead bacterial biomass was suggested to be the main driving factor responsible for elevated mineralization rates leading to oxygen depletion and consequential phosphorus release.

6. Conclusion

Drying events led to enhanced phosphorus release upon re-wetting, which could be related to enhanced mineralization rates, enhanced reduction of iron hydroxides and a general loss in sorption capacity due to increased crystallinity of iron hydroxides.

The magnitude of internal phosphorus loading upon flooding was shown to rise with the degree of sediment dehydration: The longer the intermittent drying period lasted, the higher was the rise in phosphorus release compared to previous flooding events.

Fast alternations between periods of desiccation and inundation are assumed to cause severe stress to the microbial community and thereby trigger internal phosphorus loading.

The reconnection of former isolated floodplains is going to favour fluctuating hydrologic conditions and is therefore expected to initially lead to high rates of phosphorus release from sediments that are stimulating primary production in the water bodies.

Considering longer time periods, restoration of hydrological dynamics is expected to inhibit massive eutrophication processes in reconnected sites by enabling export of nutrients and algae from the floodplain to the river and thereby inhibiting long-lasting deposition of large amounts of organic matter that are serving as a nutrient source for aquatic primary production.

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APPENDIX

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Zusammenfassung

In der vorliegenden Arbeit untersuchte ich das Phosphor-Rücklösungsverhalten von Sedimenten isolierter Augewässer östlich von Wien (Untere Lobau). Ich konzentrierte mich auf Phosphor-Dynamik, da Phosphor in Süßwasser-Systemen den limitierenden Nährstoff für die Primärproduktion darstellt. Erhöhte Phosphor-Konzentrationen begünstigen Algenblüten, die zu Sauerstoffzehrungsprozessen und im Fall von Cyanobakterien auch zur Freisetzung von Toxinen führen können.

Im Hinblick auf Restaurations-Pläne in der Unteren Lobau mit dem Ziel, Verlandungsprozesse durch die Wiederherstellung der hydrologischen Dynamik zu stoppen, ist die Reaktion von Sedimenten auf häufig abwechselnde Austrocknungs- und Überflutungsperioden ein zentraler Faktor. Man weiß bereits, dass wechselnde hydrologische Bedingungen zu Veränderungen im pH-Wert, den Redoxbedingungen, der Kristallinität von Mineralen und der Zusammensetzung und Aktivität der mikrobiellen Gemeinschaft führen. Da die genannten Parameter wichtige Einflussfaktoren für die Rücklösung von Phosphor aus Sedimenten darstellen, lag die Erwartung nahe, dass mit einer erhöhten hydrologischen Dynamik auch Änderungen in der Phosphor-Rücklösung einhergehen.

Um die Auswirkungen näher zu untersuchen führte ich ein Labor-Experiment durch, in dem Sedimente unterschiedlichen Nass/Trocken-Regimen ausgesetzt wurden. Es konnte gezeigt werden, dass die Dauer der vorangehenden Trockenperiode zu einer gesteigerten Freisetzung von Totalphosphor während der darauffolgenden Überflutungsphase führt.

Eine Partielle Korrelationsanalyse, in der die Dauer der Trockenperiode und die Probenahme-Standorte als Kovariaten berücksichtigt wurden, zeigte, dass sowohl zwischen der Änderung im Totalphosphor-Gehalt und jener im Ammonium-Gehalt, als auch zwischen der Änderung im Totalphosphor-Gehalt und jener im Eisen-Gehalt eine starke und hoch signifikante Korrelation besteht. Dieses Ergebnis könnte ein Hinweis darauf sein, dass erhöhte Mineralisationsraten und die Reduktion von Eisenhydroxiden, die zu einer gleichzeitigen Freisetzung von Eisen und Phosphor in die Wassersäule führen, dafür verantwortlich sind, dass die Phosphor-Rücklösung mit steigender Dauer der vorangehenden Trockenperiode zunimmt.

Abwechselnde Nass-/Trocken-Phasen, die jeweils 100 h dauerten und über einen Zeitraum von 600 h simuliert wurden, hatten einen signifikant positiven Effekt auf die Phosphor-Freisetzung bei Überflutung.

Die Wiederholung einer Trockenphase von 200 h und einer Überflutungsphase von 100 h über einen Zeitraum von 600 h hatte ebenfalls einen signifikant positiven Effekt auf die Phosphor-Freisetzung bei Überflutung.

Diese Ergebnisse legen den Schluss nahe, dass eine erhöhte hydrologische Dynamik anfangs zu erhöhten Phosphor-Konzentrationen in den Augewässern und damit zu einer Ankurbelung der Primärproduktion führen könnte. Betrachtet man längere Zeiträume, ist allerdings zu erwarten, dass dieser anfängliche Eutrophierungseffekt dadurch überlagert wird, dass sich das restaurierte Augebiet von einem isolierten System, das durch interne Nährstofffreisetzung und autochthone Produktion gekennzeichnet ist, zu einem mit dem Hauptfluss in Verbindung stehenden System, das auch in der Lage ist, Nährstoffe und aufgebaute Biomasse zu exportieren, entwickeln wird.

Data collection

Table A Row data of organic content determined by LOI (Loss on ignition), amorphic iron digested after Loeb et al. (2008) and different phosphorus fractions digested after Ruban et al. (2001). The iron extract was analysed with a Hitachi U-2000 photometer according to US Standard Methods 3500-Fe D and DIN 38406-E1-1. Phosphorus extracts were analysed with a Systea 3rd Generation Continuous Flow Analyser (Alliance instruments) according to ÖNORM EN ISO 15681-2. Data shown include four dry/wet cycles (A, B, C, D, see Figure 2) and three sampling sites (1 = "Schönauer Wasser 1", 2="Schönauer Wasser 2", 3="Hanslgrund", see Figure 1).

cycle	Sampling site	Replicate number	Organic content [%]	Amorphic Iron [g kg^{-1}]	Total P [mg kg^{-1}]	Organic P [mg kg^{-1}]	Inorganic P [mg kg^{-1}]	SRP [mg kg^{-1}]	Mn and Fe-bound P [mg kg^{-1}]	Ca and Mg-bound P [mg kg^{-1}]
A	1	1	8.41	7.58	574.20	170.28	403.92	0.61	372.60	395.03
A	1	2	6.68	9.80	592.89	193.97	398.92	0.74	364.59	407.87
A	1	3	10.49	11.22	615.33	210.87	404.46	0.34	364.43	401.19
A	2	1	7.45	7.93	561.16	195.33	365.83	0.77	327.29	213.31
A	2	2	5.82	8.07	554.84	149.36	405.49	1.01	360.63	317.10
A	2	3	5.49	8.53	530.76	131.48	399.28	0.85	357.86	388.55
A	3	1	18.29	3.38	566.13	156.96	409.16	2.03	390.78	531.21
A	3	2	16.02	5.25	579.26	182.23	397.03	2.88	375.51	382.37
A	3	3	21.23	3.83	623.76	215.87	407.90	2.26	391.39	476.87

cycle	Sampling site	Replicate number	**Organic content** [%]	**Amorphic Iron** [g kg^{-1}]	**Total P** [mg kg^{-1}]	**Organic P** [mg kg^{-1}]	**Inorganic P** [mg kg^{-1}]	**SRP** [mg kg^{-1}]	**Mn and Fe-bound P** [mg kg^{-1}]	**Ca and Mg-bound P** [mg kg^{-1}]
B	1	1	8.60	8.40	553.90	159.39	394.51	0.57	364.68	347.72
B	1	2	6.61	7.14	526.62	146.73	379.88	0.82	348.38	389.61
B	1	3	8.78	10.87	433.07	25.32	407.75	0.73	375.57	377.14
B	2	1	7.76	10.89	586.87	186.87	400.00	0.58	356.98	383.45
B	2	2	6.82	7.64	495.03	102.53	392.51	0.46	360.00	165.91
B	2	3	5.88	6.96	521.17	106.87	414.30	0.59	373.74	367.94
B	3	1	14.58	3.76	612.41	209.08	403.33	1.15	383.94	474.66
B	3	2	17.87	4.51	547.86	142.41	405.45	3.98	391.41	422.87
B	3	3	17.97	4.53	552.71	156.83	395.88	5.80	379.85	355.54
C	1	1	6.89	8.92	556.64	144.87	411.77	0.66	384.52	327.79
C	1	2	9.69	6.53	545.28	162.51	382.77	0.94	354.78	407.29
C	1	3	6.73	10.47	558.73	149.86	408.87	0.81	372.52	597.53
C	2	1	6.95	10.34	549.66	161.73	387.93	0.53	348.03	362.30
C	2	2	8.09	9.13	552.62	151.79	400.83	1.14	359.61	401.93
C	2	3	6.67	8.89	546.04	148.07	397.97	1.48	347.31	411.35
C	3	1	18.61	4.29	306.05	-	366.93	2.50	349.43	394.53
C	3	2	17.42	3.20	585.48	180.77	404.71	2.82	384.64	409.97

cycle	Sampling site	Replicate number	Organic content [%]	Amorphic Iron [g kg^{-1}]	Total P [mg kg^{-1}]	Organic P [mg kg^{-1}]	Inorganic P [mg kg^{-1}]	SRP [mg kg^{-1}]	Mn and Fe-bound P [mg kg^{-1}]	Ca and Mg-bound P [mg kg^{-1}]
C	3	3	7.47	3.08	478.24	90.86	387.38	0.36	374.47	426.92
D	1	1	8.76	7.81	330.81	-	405.18	1.26	380.74	399.83
D	1	2	15.63	13.83	445.02	92.22	352.80	1.04	332.75	413.00
D	1	3	13.00	10.05	512.79	138.46	374.33	0.45	328.41	398.35
D	2	1	6.47	9.00	550.21	189.09	361.13	0.71	314.99	397.22
D	2	2	6.56	8.89	609.11	224.27	384.84	1.42	336.34	405.43
D	2	3	7.17	9.58	582.25	215.81	366.44	0.48	322.50	414.64
D	3	1	21.03	4.88	552.85	185.60	367.25	5.52	341.46	436.41
D	3	2	14.70	4.38	645.39	230.17	415.22	2.00	395.11	450.64
D	3	3	19.15	4.27	581.74	222.33	359.42	2.53	337.65	407.39

Table B Row data of nutrient concentrations measured in the water column during wetting periods. Nutrient concentrations were determined according to ÖNORM EN ISO 15681-2 (PO_4^{3-}), DIN EN ISO 13395 (NO_3^- and NO_2^-) and DIN EN ISO 11732 (NH_4^+) by Continuous Flow Analysis using a Systea 3rd Generation Continuous Flow Analyser (Alliance instruments). Total Phosphorus was digested with persulfate according to Eaton & Franson (2005) and afterwards analysed as PO_4^{3-}. Concentrations of $Fe^{2+/3+}$ were determined by Lange LCK 320 cuvette test using a Hach DR-2800 portable photometer. Data shown include four dry/wet cycles (A, B, C, D, see Figure 2) with a maximum of three wetting periods (Figure 2) and three sampling sites (1 = “Schönauer Wasser 1”, 2=”Schönauer Wasser 2”, 3=”Hanslgrund”, see Figure 1).

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
A	1	1	1	100	33.4	27.6	1001.4	90.8	86.4	0.036	0.045	0.081
A	1	1	2	100	8.3	8.1	192.4	3.6	19.1	0.028	0.009	0.037
A	1	1	3	100	21.6	11.1	400.8	101.6	239	0.049	0.012	0.061
A	1	2	1	100	16.0	11.4	355.3	30.8	29	0.018	0.063	0.081
A	1	2	2	100	8.9	2	421.3	23.5	16.8	0.03	0.063	0.093
A	1	2	3	100	4.9	0.4	316.8	8.9	17.3	0.051	0.032	0.083
A	1	3	1	100	30.4	61.7	404.7	31.4	34.5	0.058	0.017	0.075

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
A	1	3	2	100	74.5	61.1	285.1	101.4	101.7	0.032	0.017	0.049
A	1	3	3	100	20.9	13.9	395.9	54.8	43.3	0.017	0.048	0.065
B	1	1	1	100	23.0	8.5	930.8	198	725	0.066	0.064	0.13
B	1	1	2	100	28.1	12.8	1192.6	250	57.6	0.068	0.029	0.097
B	1	1	3	100	80.2	20.9	33.8	6.1	5590	0.169	0.045	0.214
B	1	2	1	100	39.3	26.3	1335.2	151	160.2	0.044	0.002	0.046
B	1	2	2	100	58.4	37.3	791.3	352	1383	0.11	0.077	0.187
B	1	2	3	100	122.0	4.2	29.8	2.5	9980	0.162	0.812	0.974
B	1	3	1	100	151.3	128.7	728.8	746	998	0.078	0.103	0.181
B	1	3	2	100	62.9	47.8	864.1	60.8	31.2	0.079	0.044	0.123
B	1	3	3	100	168.8	127.1	246.2	290	4925	0.082	0.048	0.13
A	2	1	1	100	25.2	12.8	2097.8	18.6	40.3	0.011	0.034	0.045

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
A	2	1	2	100	30.6	10.1	1351.2	15.1	85.9	0.005	0.038	0.043
A	2	1	3	100	9.8	13.2	1168.1	50.5	71.9	0.012	0.034	0.046
A	2	2	1	100	9.7	3.4	1056.7	24.7	60.9	0.068	0.041	0.109
A	2	2	2	100	5.6	5	1824.5	52.5	78.8	0.037	0.09	0.127
A	2	2	3	100	18.7	2.3	1817.1	41.2	95.8	0.011	0.069	0.08
A	2	3	1	100	9.6	19.9	684.2	13.2	59.4	0.019	0.047	0.066
A	2	3	2	100	14.7	14	762.3	16.7	46.2	0.008	0.019	0.027
A	2	3	3	100	335.2	292	174.2	97	4950	0.089	0.018	0.107
C	1	1	1	100	199.2	31.4	9.1	4.5	9370	0.127	0.883	1.01
C	1	1	2	100	106.4	24.1	7.6	9.2	5740	0.031	0.075	0.106
C	1	1	3	100	170.2	6.7	9.6	6.6	6270	0.055	0.208	0.263
C	1	2	1	100	356.2	19.3	-	11	12040	0.323	2.06	2.383

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
C	1	2	2	100	292.2	69.4	-	3.4	10110	0.121	0.546	0.667
C	1	2	3	100	512.2	14.8	25.1	4.1	10030	0.256	2.26	2.516
C	1	3	1	100	398.2	257.5	1.1	8.7	14680	0.153	0.3	0.453
C	1	3	2	100	247.7	178	13	27.3	9620	0.052	0.084	0.136
C	1	3	3	100	191.2	22.4	0.3	6.9	4480	0.154	0.179	0.333
D	1	1	1	100	262.2	0.4	737.5	5	7.3	0.06	0.726	0.786
D	1	1	2	100	450.2	0.2	733.6	4.7	3.9	0.171	1.52	1.691
D	1	1	3	100	344.2	0.2	732.6	5.1	10.1	0.198	1.31	1.508
D	1	2	1	100	358.2	7.8	25.2	4.7	9630	0.301	3.43	3.731
D	1	2	2	100	582.2	15.3	18.7	6.4	17720	1.1	6.01	7.11
D	1	2	3	100	542.2	22.8	23.9	5.5	13880	0.394	2.68	3.074
D	1	3	1	100	576.2	29.2	11.5	8.2	20350	0.107	0.531	0.638

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
D	1	3	2	100	470.2	33.9	-	23.7	17320	0.094	0.57	0.664
D	1	3	3	100	858.2	10.6	30.1	9.1	13340	0.184	1.79	1.974
A	3	1	1	100	29.2	15.6	2224.1	17.3	17.9	0.039	0.086	0.125
A	3	1	2	100	178.0	50.7	-	44.1	5630	0.058	0.295	0.353
A	3	1	3	100	36.4	20.2	2472.8	105.4	53.3	0.072	0.015	0.087
A	3	2	1	100	13.9	3.3	2351.7	31.5	14.3	0.036	0.101	0.137
A	3	2	2	100	18.1	4	4848	130.5	24	0.05	0.134	0.184
A	3	2	3	100	153.8	76.6	972.3	10.4	6710	0.046	0.177	0.223
A	3	3	1	100	38.4	23.2	1926.5	33.3	10.6	0.028	0.058	0.086
A	3	3	2	100	535.7	443	108.7	165	8130	0.266	-	-
A	3	3	3	100	65.0	48.5	1437.4	52.5	9.2	0.038	0.066	0.104
B	2	1	1	100	59.4	26	885.4	248	2488	0.041	0.083	0.124

cycle	Flooding number	Sampling site	Replicate number	time [h]	P_{tot} [µg l^{-1}]	PO_4^{3-} [µg l^{-1}]	NO_3^- [µg l^{-1}]	NO_2^- [µg l^{-1}]	NH_4^+ [µg l^{-1}]	Fe^{2+} [mg l^{-1}]	Fe^{3+} [mg l^{-1}]	Fe_{tot} [mg l^{-1}]
B	2	1	2	100	141.0	47	23.5	23.9	5570	0.118	0.039	0.157
B	2	1	3	100	58.4	18.5	44.8	81	3638	0.096	-	-
B	2	2	1	100	328.2	9.2	4.8	3.7	10350	0.149	1.02	1.169
B	2	2	2	100	471.2	5.2	9.2	3.5	11210	0.232	0.71	0.942
B	2	2	3	100	177.0	82.6	36.5	40.8	8570	0.099	0.1	0.199
B	2	3	1	100	749.2	430	6.4	9.4	12280	0.12	0.273	0.393
B	2	3	2	100	730.2	322	8.5	7.1	20600	0.143	0.63	0.773
B	2	3	3	100	539.2	196	3.9	7.8	15560	0.23	0.531	0.761

Table C pH and O_2 concentration after 24h and 100h of wetting. Values shown represent means of all sampling sites pooled together.

cycle	flooding number	Sampling site	Replicate number	**time [h]**	**pH**	**O_2 [mg l^{-1}]**
A	1	1	1	24	-	-
A	1	1	2	24	-	-
A	1	1	3	24	-	-
A	1	2	1	24	-	-
A	1	2	2	24	-	-
A	1	2	3	24	-	-
A	1	3	1	24	-	-
A	1	3	2	24	-	-
A	1	3	3	24	-	-
B	1	1	1	24	7.67	4.95
B	1	1	2	24	7.94	4.40
B	1	1	3	24	7.88	2.44
B	1	2	1	24	7.98	4.42
B	1	2	2	24	7.93	3.59
B	1	2	3	24	7.87	2.50
B	1	3	1	24	7.93	4.03
B	1	3	2	24	7.87	3.62
B	1	3	3	24	8.00	4.32
A	2	1	1	24	8.16	5.38
A	2	1	2	24	8.17	5.07
A	2	1	3	24	7.93	4.34

cycle	flooding number	Sampling site	Replicate number	**time [h]**	**pH**	**O_2 [mg l^{-1}]**
A	2	2	1	24	8.16	5.61
A	2	2	2	24	8.14	5.44
A	2	2	3	24	8.13	5.79
A	2	3	1	24	8.17	4.93
A	2	3	2	24	8.04	4.43
A	2	3	3	24	7.91	4.2
C	1	1	1	24	7.64	0.35
C	1	1	2	24	7.8	1.13
C	1	1	3	24	7.79	0.57
C	1	2	1	24	7.63	0.61
C	1	2	2	24	7.77	0.84
C	1	2	3	24	7.69	0.81
C	1	3	1	24	7.76	0.61
C	1	3	2	24	7.77	0.22
C	1	3	3	24	7.8	0.55
D	1	1	1	24	7.66	0.91
D	1	1	2	24	7.59	1.18
D	1	1	3	24	7.63	0.8
D	1	2	1	24	7.66	0.22
D	1	2	2	24	7.62	0.43
D	1	2	3	24	7.65	0.59
D	1	3	1	24	7.66	0.37
D	1	3	2	24	7.85	0.4

cycle	flooding number	Sampling site	Replicate number	time [h]	pH	O_2 [mg l^{-1}]
D	1	3	3	24	7.69	0.73
A	3	1	1	24	8.2	5.06
A	3	1	2	24	7.86	0.7
A	3	1	3	24	8.09	4.72
A	3	2	1	24	8.16	5.46
A	3	2	2	24	8.25	5.8
A	3	2	3	24	7.8	1.06
A	3	3	1	24	8.04	4.61
A	3	3	2	24	7.77	0.36
A	3	3	3	24	7.97	4.15
B	2	1	1	24	8.1	4.87
B	2	1	2	24	7.98	2.78
B	2	1	3	24	7.99	4.37
B	2	2	1	24	7.78	0.59
B	2	2	2	24	7.95	2.98
B	2	2	3	24	8	4.27
B	2	3	1	24	7.84	0.47
B	2	3	2	24	7.7	0.52
B	2	3	3	24	7.83	0.28
A	1	1	1	100	-	-
A	1	1	2	100	-	-
A	1	1	3	100	-	-
A	1	2	1	100	-	-

cycle	flooding number	Sampling site	Replicate number	time [h]	pH	O_2 [mg l^{-1}]
A	1	2	2	100	-	-
A	1	2	3	100	-	-
A	1	3	1	100	-	-
A	1	3	2	100	-	-
A	1	3	3	100	-	-
B	1	1	1	100	8.03	5.62
B	1	1	2	100	7.91	4.86
B	1	1	3	100	8.02	4.63
B	1	2	1	100	7.98	4.76
B	1	2	2	100	8.08	5.21
B	1	2	3	100	7.95	0.2
B	1	3	1	100	7.99	4.03
B	1	3	2	100	7.86	4.19
B	1	3	3	100	7.9	3.63
A	2	1	1	100	8.1	5.44
A	2	1	2	100	8.1	5.22
A	2	1	3	100	7.91	4.72
A	2	2	1	100	8.06	5.27
A	2	2	2	100	8.09	5.58
A	2	2	3	100	8.04	5.78
A	2	3	1	100	8.19	4.87
A	2	3	2	100	8.07	4.56
A	2	3	3	100	8.04	3.92

cycle	flooding number	Sampling site	Replicate number	time [h]	pH	O_2 [mg l^{-1}]
C	1	1	1	100	8.01	0.1
C	1	1	2	100	8.17	2.7
C	1	1	3	100	8.13	1.39
C	1	2	1	100	8.03	2.89
C	1	2	2	100	8.08	4.48
C	1	2	3	100	7.99	5.76
C	1	3	1	100	8.11	4.2
C	1	3	2	100	8.33	5.65
C	1	3	3	100	8.26	5.3
D	1	1	1	100	7.93	2.08
D	1	1	2	100	7.81	0.23
D	1	1	3	100	8.12	3.2
D	1	2	1	100	7.86	0.28
D	1	2	2	100	8.01	0.11
D	1	2	3	100	8.03	0.1
D	1	3	1	100	8.27	3.06
D	1	3	2	100	8.06	1.7
D	1	3	3	100	8.01	1.79
A	3	1	1	100	8.16	6.02
A	3	1	2	100	8.14	4.55
A	3	1	3	100	7.93	4.83
A	3	2	1	100	8.08	5.8
A	3	2	2	100	7.99	5.67

cycle	flooding number	Sampling site	Replicate number	time [h]	pH	O_2 [mg l^{-1}]
A	3	2	3	100	8.06	5.59
A	3	3	1	100	7.99	5.92
A	3	3	2	100	8.07	4.27
A	3	3	3	100	7.96	5.03
B	2	1	1	100	8.02	5.21
B	2	1	2	100	8.24	3.87
B	2	1	3	100	7.92	3.2
B	2	2	1	100	8.12	0.12
B	2	2	2	100	7.83	0.29
B	2	2	3	100	7.99	4.73
B	2	3	1	100	8.08	2.58
B	2	3	2	100	7.99	1.75
B	2	3	3	100	8.07	2.79

Curriculum vitae

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education:

1993 – 1997 elementary school in Vienna
1997 – 2005 secondary school in Vienna
June 2005 general qualification for university entrance (passed with distinction)
2005 – 2011 Studies of Biology at the University of Vienna
2005 – present Studies of Law at the University of Vienna

Work experience:

July 2005: Placement at the Medical University of Vienna (Institute of pharmacology) in context of the "Gen-au Summer School"
July 2006: Placement at the urban administration of the City of Vienna (Department of Environmental Protection)
March 2007 – Jan 2008: assistant in the context of a project dealing with the toxicity of *Bacillus thuringiensis israelensis* on Large Branchiopods
Aug 2008: Placement at the urban administration of the City of Vienna (Department of Environmental Protection)

Scholarships:

2008: Performance Scholarship offered by the Federal Ministry of Science, Art and Culture
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2010: Research Scholarship for diploma thesis in Ecology offered by the Federal Ministry of Science, Art and Culture

Publications and Conference Contributions:

I.M. Schönbrunner, P. Preiner & T. Hein, 2011. Impact of drying and re-flooding of sediment on the phosphorus dynamics of river-floodplain systems. Conference on the Status and the Future of the World's Large Rivers, Vienna, April 11-14, 2011.

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I. M. Schönbrunner & E. Eder, 2006. pH related hatching success of *Triops cancriformis* (Crustacea: Branchiopoda: Notostraca). Hydrobiologia 563: 535-538.

E. Eder, C. Strondl & I. M. Schönbrunner, 2006. Natural and anthropogenic barriers protect populations of the stone crayfish (*Austropotamobius torrentium*; Decapoda: Astacidae) near Vienna. 4th European Conference on Biological Invasions, Vienna