MASTERARBEIT / MASTER'S THESIS

Titel der Masterarbeit / Title of the Master's Thesis

„Effects of riverbed restoration on total transient storage and nutrient spiraling of an urban river: an experimental solute addition study on River Wien (Vienna, Austria)"

verfasst von / submitted by

Birgit Goernet, BSc

angestrebter akademischer Grad / in partial fulfilment of the requirements for the degree of

Master of Science (MSc)

Wien, 2017 / Vienna 2017

Studienkennzahl lt. Studienblatt / degree programme code as it appears on the student record sheet:	A 066 833
Studienrichtung lt. Studienblatt / degree programme as it appears on the student record sheet:	Ecology and Ecosystems UG2002
Betreut von / Supervisor:	emer. o. Univ.-Prof. Dr. Friedrich Schiemer

AKNOWLEDGEMENTS

This research was carried out as part of the River Wien monitoring programme: ‘Hydro-chemistry, suspended solids dynamics and organic pools in consideration of the flood-protection function of the River Wien retention basins’. I acknowledge the support of and funding by the Municipal Water Authority of Vienna and the Federal Ministry for Agriculture and Forestry, Environment and Water Management.

At this point I would like to express my gratefulness towards Univ. Prof. Dr. Thomas Hein and lecturer Dr. Gabi Weigelhofer for their professional guidance, ideas and enthusiasm for this master thesis: Thomas for giving me the opportunity to conduct this research and for the preliminary compilation of the work design, Gabi for the enormous support on data analysis. Both I want to thank for the relaxed and amiable working atmosphere, always letting me feel comfortable to address questions.

Emer. o. Univ. Prof. Fritz Schiemer acknowledgements of gratitude for the interest in this research work and having challenged me as a “sparring partner”! I am deeply grateful for the time offered, valuable comments on drafts and for the huge patience.

Sincere thanks goes to the Department of Limnology and Oceanography for providing laboratorial facilities as field sampling equipment. A special thank you goes to the Austrian EU Stream Expert System working group for having imparted the slug-addition method and Hubert Kraill for the introduction to the world of chemical water analysis through very strict, but extraordinary competent working instructions. My thanks are also extended to Stefan Glaser for first advices in fieldwork and to Christina Marchesani for helping me drag up to fifty liter of injection solution through water and reed. Furthermore I would like to point out my appreciation towards all the great ladies Andrea, Birgit, Irene and Anna in the office room for exchanging literature-search as well as software skills and advices in scientific writing.

A profound thank you goes to my former project colleagues Horstl and Mikl, as to the associated ‘electrofishing family’, for repetitively letting me know: “just finish”.

Dear mum and dad, I appreciate you respecting my decision to go for the field of biology. Mum, thank you for joining me in river morphology measurements. Dad, thank you for all the arduous work to support the family and co-financing my studies. Brother Ronny thank you for taking sediment cores with me even on the weekend.

Flat mates and motivation heroines Miguel, Mimi and Mary observing the three of you spending months of brainwork on your PhD taught me necessity of continuous focus. Dance Gang, thank you for all the encounters on the dance floor. Heartfelt gratitude, Grandmother Oya. G.G., coyote teacher, I bow to you midst closing this cycle.

In conclusion, Sabien, Ingrid, Karin, Marlen, Armin, Thomas, Fabio, Stefan, Sarah, Isa, Deli, Julia, Ka Ayra, Staffi and Sue the biggest hug to you for accompanying me throughout my studies, in this more than a decade time keeping my spirits up and reminding me that the effort pays off when all the work finally is accomplished.

TABLE OF CONTENTS

ABSTRACT

Effects of riverbed restoration on total transient storage and nutrient spiraling of 4th order River Wien at the western city limits of Vienna (Austria) were examined. Focus was to gain deeper insight on riverine internal structures and processes of two restructured flow-sections different in rehabilitation measures and thus in ecological conditions complementing the compiled data of the monitoring project "Hydro-chemistry, suspended solids dynamics and organic pools of the River Wien". For this task a solute addition experiment (SAE) at the reach-scale was conducted during low flow (0.33 $m^3 s^{-1}$) conditions. Three slug-type injections of the conservative tracer NaCl and the nutrients P (as Na_2HPO_4) and N (as NH_4Cl) were realized in July / August / September 2002. Tracer and nutrient data were fit to a one-dimensional transport and storage model. Supplemental a heavily modified section was studied to grasp potential opportunities for matter retention.

The restructured reach II (reactivated in-stream wetland) was found the most geo-morphological heterogeneous section and exhibited largest total transient storage. Channel complexity (relative sum of sinuosity, variance of river width such as depth and sediment sorting-coefficient) estimate was the twofold of reach I (revitalized riffle-pool) and the fourfold of reach III (channelized run). Overall variation of width and depth as well as the amount of fine sediments controlled transient storage. New structural conditions resulted both in a distinct slowdown of flow velocity and in an increase of water residence time. 34 – 51 % of River Wien calculated cross-sectional areas were identified as transient storage zones (*TSZs*, compartments delaying the longitudinal water downstream transport). At reach II the water remained in the *TSZ* 1.6 and 2.2 times longer compared to reach I and III. Furthermore water residence time in the *TSZ* contributed stronger to transient storage than *TSZ* size did.

Tendencies for the spiraling of P identified reach I above II holding higher capacities for and performing faster in biogeochemical uptake. Transient storage seemed not an important driver for the spiraling of P at River Wien restructured sections. Likewise no correlations of spiralling metrics with geomorphological factors could be found. Sediment data although indicated P having been to most parts chemically adsorbed to fine-grained sediments. Mass balances revealed N over P retained in the restructured sections. At reach III, on the contrary, transient water and solute storage zones had impact on the uptake efficiency of P. S_W (uptake length) was correlated with A_S/A (ratio between the cross-sectional area of the free flowing water-column and the cross-sectional area of the *TSZ*) as well as with *TSZ* extent.

The restoration of riverbed physical characteristics resulted in an improvement of River Wien structural and functional traits. Attained knowledge on linkages amongst geomorphic attributes and total transient storage particularly may inform on potentials in river rehabilitation respectively on priorities in hydraulic engineering.

INTRODUCTION

Lotic ecosystems are a fundamental landscape feature supporting wetland biodiversity, regulating the hydrologic balance and providing vital ecological goods and services (Palmer *et al.* 2004; Bukaveckas 2007). They play a major role in the recharge of the groundwater body, processing of nutrients and purification of water (Richardson 1994). The functional ecosystem traits of streams with sound condition have a great impact on global biogeochemical cycles and at the same time are of central value to society for flood protection and eutrophication control (Jungwirth *et al.* 2002; Sullivan *et al.* 2004; Hein *et al.* 2005; Loreau 2008).

The river channel provides a flume for runoff and nutrient export from the watershed to downstream ecosystems (Minshall *et al.* 1983; Seitzinger *et al.* 2002). Channel morphology and sediment composition determine the speed of water while grain-sizes and quality of the sediment provide the biogeochemical active surfaces. The bed-structure and the inherent biocoenosis alter the timing, form and magnitude of matter movement through both abiotic and biotic processes (Meyer *et al.* 1988; Burns 1998; Alexander *et al.* 2000). Bifurcating pristine rivers, encompassing the riparian ecotone during flooding, display lateral – transversal – vertical connectivity and large submerged surfaces. Such structural conditions exhibit extended transient water and solute storage-zones resulting in an increase of water residence-time, particle deposition and nutrient retention (Stofleth *et al.* 2008).

The delay in longitudinal water transport is referred to as transient storage (Bencala & Walters 1983). As a response to friction, recirculation and down-welling, parts of the waterbody flow slower than the average flow velocity (Runkel et al. 2002). River pools, backwater, debris dam as well as the sediment porous-space of the hyporheic zone function as transient storage-zones (*TSZs*) (Thackston & Schnelle 1970; Bencala & Walters 1983; Ensign & Doyle 2005; Stofleth *et al.* 2008; Bottacin-Busolin *et al.* 2009). *TSZs* are important physical templates for nutrient dynamics in rivers and drive material processing (Fisher *et al*. 1998; Peterson *et al.* 2001; Battin *et al.* 2003; Gooseff *et al.* 2007). A long contact time of dissolved nutrients with reactive sites results in higher rates of adsorption, complexation, biotic uptake and conversion thereby controlling the geochemical and biological retention of nutrients (Kim *et al*. 1992; Valett *et al*. 1996; Butturini & Sabater 1999; Hall *et al.* 2002; Craig *et al*. 2008; Stofleth *et al*. 2008; Mulholland & Webster 2010).

Nutrient precipitation, sorption-desorption reactions and biological uptake-remineralisation processes (Allan 1995) take place at the seston particles, at the benthic and hyporheic zone. Essential elements such as carbon, phosphorus and nitrogen are incorporated into and regenerated from living tissue in a cycle (Newbold 1992). Biological nutrient retention is reflected in the abundance of free-living and attached bacteria, primary and secondary production. The number of free bounding sites in complex-compounds, the pH-value and changes in sediment redox-potential are the triggering factors concerning geochemical nutrient cycling. Consequently the rates of biogeochemical nutrient processing are dependent on the transient storage, light, temperature, water chemistry, substrata and riverine communities.

Fluvial networks up to the mid 1970s were considered primarily as transport conduits lacking mentionable dynamics of particles and solutes (Howarth *et al.* 1996; Mulholland & Webster 2010). The Nutrient Spiraling (Webster 1975; Wallace *et al.* 1977; Webster & Patten 1979) and the River Continuum Concept (Vannote *et al.* 1980) opened up the view of the river towards an in-stream matter processing and downstream orientated transporting system (Newbold *et al.* 1982). Each digestion cycle is displaced downward from the previous in a spiraling motion (Allan & Castillo 2007). Continuous research on nutrient dynamics showed considerable uptake and transformation of N and P within and pointed out the highly retentive nature of rivers (Newbold *et al.* 1983a; Mulholland 1992; Newbold *et al.* 1992; Hall 2003; Bernhardt *et al*. 2005a). Interest in impact of urbanization on ecosystem function and basin-scale matter retention has perked up research on nutrient spiraling (Mulholland & Webster 2010).

Solute retention-transport models aim to clarify the key factors of influence and the co-player impact-hierarchy on the course of matter. Transient-storage and nutrient-spiraling measurements enable the estimation of hydrologic, phys- and biogeochemical river-properties (Stream Solute Workshop 1990). In addition they support the analysis of spatial and temporal solute cycling-patterns and provide information on active hot spots and moments of matter processing (Allan 1995; McClain *et al*. 2003). Newbold *et al.*, 1981, presented a mass-balance based framework-model to estimate the capacities, the velocities and the efficiencies of matter processing (Bunn *et al*. 1999). Spiraling metrics assist understanding how the river bed-structure and communities shape the flow of water and affect nutrient withdrawal (Newbold *et al*. 1982). Solute retention-transport models are utile for evaluating human impacts and quantifying the effects and benefits of restoration measures on the performance of a river (Boulton 1999; Bernhardt *et al*. 2005b).

Settlement, industrial and agricultural area-sprawl has changed channel-profiles and flow-regimes of an increasing number of streams and rivers worldwide (Groffman *et al*. 2003; Meyer *et al.* 2005; Geurts *et al*. 2008). Modifications were enforced for the benefit of land accrual, flood-protection aims, drinking- and irrigation-water supply or hydropower generation. Within urbanized catchments impermeable surfaces channel runoff into running waters fast, unfiltered and with elevated frequency and amplitude (Paul & Meyer 2001). Consequences of the disruption of the lotic continuum, disconnection from groundwater bodies and loss of floodplain or riparian areas are hydrologic flashiness, poor water quality, habitat fragmentation, low species numbers and land degradation (Poff *et al*. 1997; Paul & Meyer 2001; Meyer *et al*. 2005; Walsh *et al.* 2005; Dudgeon *et al*. 2006; Bukaveckas 2007). The diminution in ecosystem structure and function is a major environmental problem and of socio-economic disadvantage.

Sustainability of lotic ecosystems requires alleviation of anthropogenic pressures and significant improvement of the ecological status of regulated streams and rivers (Geurts *et al.* 2008). Ecologists have gained support for this ambition from the general public (Boulton 1999), water authorities as well as through statutory provision (e.g. European Water Framework Directive 2000/60/EEC). Well conceptual designed and well-implemented restoration projects necessitate interdisciplinary cooperation and the consideration of diverse stakeholders needs and uses of ecosystem services (Groffman *et al*. 2003; Palmer *et al*. 2007). Restoration practices target to initiate ecosystem recovery and to foster ecosystem resilience.

A large-scale restoration programme of the River Wien (approx. 4 km) was initiated within the city limits of Vienna in the late 1990ies. Flood protection, bank reinforcement and urbanization had caused major alterations in the river ecology (Konecny et al. 2002, p. 9). The first objective was to reconnect the main channel with the wetlands of the flood-control reservoirs (FCRs). The creation of an urban semi-natural fluvial landscape, at the mesoscale, aimed to enhance nutrient dynamics and amplify habitats for biota. The second objective was to revitalize heavily modified flow sections sealed with bottom-plates and concrete. Remediation measures consisted in withdrawal of those flagstones, replacement with sediments and riprap, as stabilization element. The intention was to increase the vertical connectivity and lateral extension, as well as to improve the physico-chemical and habitat-heterogeneity status of the river. A complementary aim was to create a new area for recreation. To comprehend the outcome of the restoration efforts an accompanying multidisciplinary monitoring project was carried out.

The purpose of the present master thesis was to contribute to a better understanding of the effects of riverbed restoration on hydro and solute dynamics of River Wien. For this objective a solute addition experiment (SAE) study was realized. Designed reaches selected for the estimation of total transient water-storage and spiralling of phosphate and ammonium were part of a restructured riffle-pool and reactivated wetland flow-section. A heavily modified section was examined in addition for comprehension of potential prospects for matter retention. Findings should add and serve comparison to mass balances data of the monitoring project ‘Hydro-chemistry, suspended solids dynamics and organic pools in consideration of the flood-protection function of the River Wien retention basins: influence of the restored retention basins VI and VII on nutrient dynamics of River Wien’ in 2002.

Study site

The study sites were located in the River Wien (Wienfluss, in German) at the western outskirts of the city of Vienna, Austria (N 48° 12,161´, E 16° 14,329´; Fig. 1). The river is characterized through a pluvionival hydrological regime, has a total length of 34 km and drains a catchment of 230 km^2 (Aschauer *et al*. 2006, p. 24–27). The spring is found in the western Vienna Woods 520 m above sea level (Web[1]). Headwater and middlecourse are situated in the alpine- and the lowercourse in the Hungarian lowland-ecoregion (Illies 1978, modified; Wimmer & Moog 1994; Aschauer *et al*. 2006, p. 24–27). The River Wien discharges to the Danube Canal, an anabranch of the River Danube, at the city center of Vienna 150 m ASL (Ladinig *et al.* 1999, p. 7; Wimmer & Moog 1994). The catchment holds 75 % rural and 25 % urban area. The rural area consists to more than two thirds of deciduous forests. Bedrock geology is part of the alpine and flysch sandstone-zone (Aschauer *et al*. 2006, p. 24–27). Viennese sandstone is characterized through fine grained and densely packed calcareous sediments orginated from marine deposition (Ladinig *et al*. 1999, p. 7) resulting in a very low pore volume and water retention-capacity. High surface runoff during rainstorms leads to formation of torrents and an instant rise in discharge (Keckeis *et al*. 2008). The longterm hydrograph is flashy. Base flow averages 0.2 $m^3\ s^{-1}$ and storm flows ($HQ_{1,000}$) peak in more than 450 $m^3\ s^{-1}$ (Web[1]). The average discharge at Vienna-Stadtpark is 1.2 $m^3\ s^{-1}$ (Aschauer *et al*. 2006, p. 24). The annually predicted flood-event discharge HQ_1 averages 14 $m^3\ s^{-1}$ (Web[1]). At base flow the Wien appears a stream, but high peaks called for classifying it a river. The phytobenthic trophic state of River Wien at Vienna-Hadersdorf is classified meso-eutrophic (after Vollenweider & Kerekes 1982) and the saprobic index is 2 to 2.5 (Wassergüte in Österreich 2010, annex; Aschauer *et al*. 2006, p. 27). Elevated average concentrations of total nitrogen (N > 2 mg L^{-1}), for the most part dissolved inorganic N, points at human influence (Dodds *et al*. 1998; Lewis *et al*. 1999; Hagedorn *et al*. 2000). The Mauerbach, a 4^{th} order stream, is the main tributary to the River Wien (Fig. 1). It contributes to discharge with 0.25 $m^3\ s^{-1}$ and its saprobic index is a two (Aschauer et al. 2006, p. 26).

The exceptionally high flood events, overall the possible 2000-fold increase in discharge within hours, as well as spreading urban settlements called for a major regulation scheme of the River Wien 120 years ago. The construction of a chain of six flood-control reservoirs (FCRs) separated from the following with concrete dams, in Vienna-Hadersdorf, was realized. With an impoundment volume of 1,160,000 m^3 (Web1) and an areal extension of 0.37 km^2 they were constructed to fill successive during a flood event (Ladinig et al. 1999, p. 9; Pic. 1). The stream Mauerbach retention basin (FCR 1) contributes to flood-control with an impoundment volume of 150,000 m^3 (Web[1]). In addition the Wienerwaldsee, a water protection area and reservoir in the Vienna woods upstream of the FCRs, adds to flood-control (Fig. 1). In further sequence approximately half of the river was channelized in terms of the technical design of the 19th century (Hager 2002, p. 10; Konecny *et al.* 2002, p. 5). These flow-sections have a straightened course, a sealed riverbed and a u-profile. Originally 124 brooks and ditches poured in to the River Wien. Thereof 11 discharged in to the river within the city boundaries. Few brooks remained as open flumes. Most are now integral part of the Viennese canalisation system (Ladinig *et al*. 1999, p. 7).

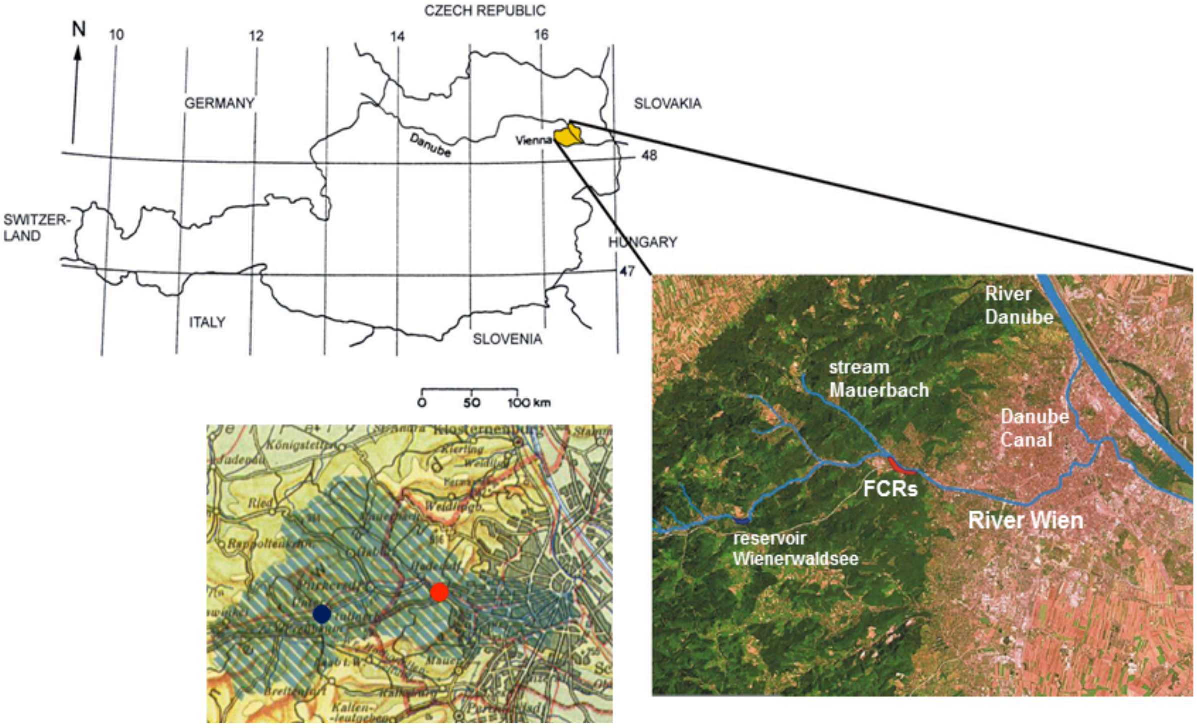


Figure 1 Location and catchment (left below; © Ed. Hölzel, Vienna) of the River Wien in Vienna, Austria. The flood-control reservoirs (FCRs) are highlighted in red. The dots in the catchment picture illustrate where the FCRs (red) and the reservoir Wienerwaldsee (dark-blue) are situated.

Pic. 1 River Wien FCR 6 (left); Flooding of the FCRs 4 and 5 (right) © not traceable (printed in: Ladinig *et al.* 1999, p. 22).

The partial restoration of the River Wien started in 1998. The integration of an approximately 1.3 km river-stretch into the FCRs called for the installation of controllable floodgates. Prior to the restoration the river bypassed the FCRs. Figure 2 shows the expansion of water surfaces after restoration and figure 3 gives a more detailed picture of the FCR wetlands. The weirs are open up to a flow of 30 $m^3 s^{-1}$. Discharge greater 30 $m^3 s^{-1}$ has an average occurrence every two years. Further excessive silt was excavated, a flow-channel was trenched and gravel and cobble were put out. The resulting hydrological connectivity of the River Wien with the FCRs allows river dynamics within this reactivated man-made wetland. Since 2001 the River Wien flows through all six FCRs. Now the FCRs represent the only remnant of the bifurcation zone of the former floodplains and are refugium for aquatic wildlife. A further rehabilitation action implemented was the restructuring of a sealed flagstone-concrete section (approx. 2.7 km) upstream of the FCRs. Restoration included the removal of the impermeable surface-cover and bank reinforcements. The reopened riverbed was filled with gravel. Riprap was put out to moderate sediment shifting and to assure riverbed stabilisation. The target was to raise the sinuosity, increase vertical and lateral exchange-processes, ameliorate the water quality and create habitats – all in order to amend the ecological status of this river section.

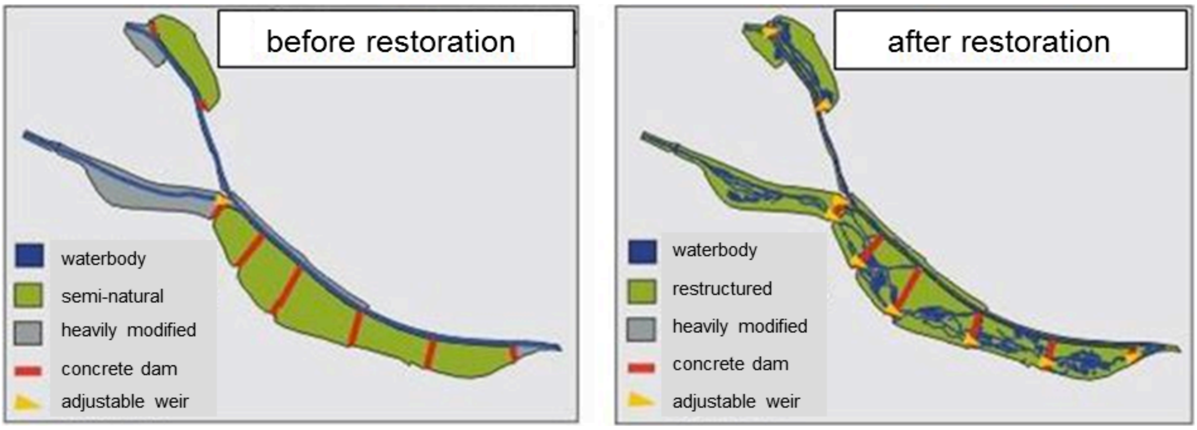


Figure 2 Schematic drawing of the River Wien and the stream Mauerbach flood-control reservoirs before and after restoration © H. Keckeis, modified.

Figure 3 Drawing of the solute addition experiment (SAE) study reaches I – III, the flood-control reservoirs of the River Wien (FCR 2-7) and stream Mauerbach reservoir (FCR 1) after restoration (© Alfons Oberhofer 2002, p. 14, modified). At the shown segment three tributaries join the River Wien. The brook Rotwassergrabenbach enters at FCR 5 and the Grünauerbach at FCR 6. The blue star on the right indicates the point of conflux of the River Wien and the stream Mauerbach. White dots give an overview of prevalently monitored water chemistry and suspended solids sampling sites.

Criteria for the selection of study reaches were that sections presented a roughly straight course and no influent flows. These requirements for achieving good mixing of solutes added and avoiding lateral dispersion were an easement of not having to take into consideration influent matter and discharge. Reaches I and II were different in restoration measures, but nearly identical in hydrology – a necessity for direct comparison. Both restructured flow-sections were situated in the 4th flow-order segment (Aschauer *et al.* 2006, p. 27). Reach III showed a channelized riverbed and was located below the FCRs and the conflux with the stream Mauerbach at the top of the remaining heavily modified segment. At this site the River Wien is of 5th flow order (Fink *et al.* 2000, p. 34). Licit comparisons with the restructured reaches considered altered hydrology and were exerted with all parameters independent from discharge. Stream Mauerbach water chemistry did not effect the SAE. Figure 3 shows the location of the study reaches in the River Wien system.

Reach I (Pic. 2) was part of the revitalized, formerly sealed and straightened, river-section upstream of the River Wien FCRs. The achieved state shows a riffle-pool sequence. Sediments consisted in large part of middle-sized gravel. Large cobble was scattered observed. The riparian vegetation was mainly made up of herbs. Full-grown willow trees were scarce, but several few-year old were emerging.

Pic. 2 Reach I (riffle-pool revitalized section) of the solute addition experiment in the River Wien.

Reach II (Pic. 3) was part of the reactivated wetlands of the FCRs. It stretched nearly across the whole FCR 6. The tributary Grünauerbach, discharging into the reach, carried no water during SAE-sampling-dates. Reach II also showed a riffle-pool sequence. The sediments comprised a mix of cobbles, sand, silt with anoxic layers and patches of clay. The riparian vegetation consisted to 80 % of reeds (Hager 2002, p. 10). Black alder and willow trees were point distributed.

Pic. 3 Reach II (reactivated wetland section) of the solute addition experiment in the River Wien.

Reach III (Pic. 4) was part of the heavily modified constrained flow section downstream of the FCRs. The riverbed at this site was sealed with flagstones and concrete and the cross section is trapezoidal. During long low-water periods bottom layers were densely covered with filamentous green-algae.

Pic. 4 Reach III (channelized run section) of the solute addition experiment in the River Wien.

Table 1 Listing of water chemistry and suspended solids sampling sites in the River Wien system. Shown are sampling sites relevant for the Solute Addition Experiment (SAE) and the monitoring project "Hydro-chemistry, suspended solids dynamics and organic pools of the River Wien" in general. Variations in start and finish of sampling at the various sites resulted from monitoring needs and construction during the phases of river restoration and building of the HLAG-train tube beneath flood-control reservoir (FCR) 6 and 7. Rotwassergrabenbach and Grünauerbach are brooks discharging into River Wien after heavy rainfall. The stream Mauerbach is the main tributary to River Wien. Hadersdorf-Weidlingau, Auhof and Hütteldorf are areas in the 13th and 14th Viennese district.

River Section	**Study sites**	**SAE**		**Monitoring**	
	SAE and Monitoring Project	**Beginning**	**Ending**	**Beginning**	**Ending**
River Wien					
revitalized riffle-pool	**Reach I, top**; Hadersdorf-Weidlingau, Bahnstraße	28.07.2002	17.09.2002	22.04.2002	04.10.2004
revitalized riffle-pool	Hadersdorf-Weidlingau, 200 m above Badgasse			04.01.1999	17.10.2005
revitalized riffle-pool	**Reach I, bottom**; Hadersdorf-Weidlingau, Badgasse	28.07.2002	17.09.2002	22.04.2002	07.10.2002
reactivated wetland	Auhof, FCR 3, 10 m below the weir			04.01.1999	17.10.2005
reactivated wetland	Auhof, FCR 4, 10 m below the weir			04.01.1999	17.10.2005
reactivated wetland	Auhof, FCR 5, 10 m below the weir			12.03.1999	17.10.2005
reactivated wetland	**Reach II, top**; Auhof, FCR 6, 10 m below the weir	28.07.2002	17.09.2002	07.03.2000	17.10.2005
reactivated wetland	**Reach II, bottom**; above conflux with the Grünauerbach	28.07.2002	17.09.2002	11.03.2002	17.10.2005
reactivated wetland	Auhof, FCR 7, 10 m below the weir			01.08.2001	17.10.2005
channelized run	Hütteldorf, 50 m below conflux with the Mauerbach			04.01.1999	17.10.2005
channelized run	**Reach III, top**; Hütteldorf, Bergmillergasse	28.07.2002	17.09.2002	22.04.2002	04.10.2004
channelized run	**Reach III, bottom**; Hütteldorf, Brudemanngasse	28.07.2002	17.09.2002	22.04.2002	07.10.2002
Rotwassergrabenbach	Auhof, FCR 5, 15 m before conflux with the River Wien			12.03.1999	23.05.2005
Grünauerbach	Auhof, FCR 6, 15 m before conflux with the River Wien			25.03.2002	23.05.2005
Mauerbach	Auhof, Utendorfgasse			04.01.1999	17.10.2005

Questions of research

I. What are the contributions of riverbed restoration on channel complexity and total transient water-storage at selected restructured reaches of River Wien and does comparison of the revitalized riffle-pool and the reactivated wetland with the channelized section, indicate an improvement, respectively a potential for further management options?

II. Aspects of nutrient spiraling – following questions were considered:
(i) Do the differently restored river-sections show distinct spiraling findings on the example of phosphate and ammonium?
(ii) Are the uptake capacities during elevated nutrient conditions more stimulated at the reactivated wetland than at the revitalized riffle pool section?
(iii) In particular, is transient storage an important driver for nutrient spiraling?

III. How do the results from the solute-additions cope with the mass-balance calculations: is the estimation of spiraling metrics worthwhile the effort and does the detailed insight offer substantial information for management concepts?

MATERIALS AND METHODS

Hydrology

Flow velocity was measured with the portable flowmeter Flo-Mate 2000™ (Marsh-McBirney, Inc.) at the reach bottom at every SAE date. Discharge (Q) was calculated according to Gore (1996). To avoid strong influence of close shore slow flow-velocities a reach transect was divided into discharge-calculation cells of 0.5 m width. Water velocity was measured in the middle of each cell at a height of 40 % above ground.

$$\text{Discharge, Q}\left(\mathrm{m^3\,s^{-1}}\right) = \frac{\sum\left(\text{width}_{\text{cell}}\left(\mathrm{m}\right) * \text{depth}_{\text{cell}}\left(\mathrm{m}\right) * \text{velocity}_{\text{cell}}\left(\mathrm{m\,s^{-1}}\right)\right)}{n_{\text{cells}}}$$

eq. 1

Within the framework of the monitoring project flow velocity was recorded at hydrologic in- and output sites of the FCRs with an Ott-Nautilus (C2000 nbr. 119736) and converted to discharge. Measurements took place at monitoring dates during the vegetation period and low to average water levels. Data was provided by the Municipal Water Authority of Vienna (MA 45) and used for calculating mass balances. The permanent gauge Pegel-Hadersdorf recorded hydrological data of the River Wien automatically in five-minute intervals. It was set up 200 m upstream of FCR 2.

Riverbed structure

Riverbed-morphology and sediment-composition data were collected for reach description, estimation of channel complexity and further discussion of transient-storage and nutrient-spiraling results.

Channel morphology

River width was measured at each reach in 20 m transect-intervals. River depth was recorded in 0.25 m spacings along each transect. Measurements for the estimation of the sinuosity (ratio of the shoreline length and the reach length) as well of the water-surface area were realised via dGPS-surveying (Pathfinder Pro XL, Trimble Navigation Limited). Data for each above-mentioned parameter was recorded once.

$$\text{Sinuosity} = \frac{\left(\text{reach perimeter}\left(\text{m}\right)*0.5\right) - \text{reach cross section}\left(\text{m}^2\right)}{\text{reach length}\left(\text{m}\right)}$$

eq. 2

Sediment composition

Superficial sediment-composition was characterised visually once at each 20 meter transect. The transect strip surveyed was 0.5 m wide. Choriotop categories were classified according to Moog (1990). Grain-sizes were recorded in percent per square meter. The grain-sizes percentiles (*Q* 25/50/75) were calculated using the programme Graintest-V1.06C-Retsch.

The sorting coefficient (S_o) was calculated after Trask (1932). S_o is an indicator for the aggregation, or dispersion, of sediment grain-sizes. 'Wellsorted' and 'scarcely dispersed' indicates sediment distribution of similar size (Kondolf *et al.* 2003, p. 351). S_o decreases with an increase in sorting.

$$\text{Sorting coefficient}, S_o = \left(\frac{Q75}{Q25}\right)^{0.5}$$

eq. 3

The skewness coefficient (S_k) was calculated after Krumbein & Pettijohn (1938). S_k gives the ratio of fine- to coarse-grained sediment. Log-transformed distributions, in general, are negatively skewed (Kondolf & Wolman 1993). S_k decreases with an increase in fine sediments. Fine sediment dominance shows a left and coarse sediment dominance a right skewed curve from a log-normal distribution.

$$\text{Skewness coefficient}, S_k = \log\left(\frac{Q25*Q75}{Q50^2}\right)$$

eq. 4

To comprise the biotic fraction of the sediments, contents of organic matter and chlorophyll *a* (Chl-*a*) were quantified and periphyton growth was estimated. Organic matter was combusted at 500 °C. Chl-*a* was extracted in 90 % acetone (samples were stored overnight refrigerated and additionally treated with ultrasound), measured spectro-photometrically with a Hitachi U-2000 and calculated according to Parsons *et al.* (1984). Wavelength extinctions (E) were corrected by the 750 nm acetone blank reading.

$$\text{Chl-a}\left(\mu\text{gL}^{-1}\right)=\frac{\left(\left(11.85*\text{E}_{664}\right)-\left(1.54*\text{E}_{647}\right)-\left(0.08*\text{E}_{630}\right)\right)*\text{V}_{\text{acetone}}\left(\text{mL}\right)}{\text{V}_{\text{sample}}\left(\text{L}\right)*\text{light path cuvette}\left(\text{cm}\right)}$$

eq. 5

In both reach I and II ten sediment samples (depth 0 – 0.05 m) of each present choriotop category, with exception of cobbles > 200 mm, were randomly taken once during the vegetation period of 2002. Sediment from silt to pebbles I was collected with a PVC core (diameter = 6 cm). Grain-size classes greater 20 mm were gathered using the Hess-sampler (diameter = 25 cm).

In reach III only periphyton samples could be taken. Periphyton was scraped off the flagstones, 100 cm^2 respectively. The areal expansion of the periphyton (%) was estimated visually.

The average standing-stocks of organic matter and chl-*a* were weighted according to the respective grain-size composition of each reach. Calculation results refer to 1 m^2 sediment surface and 0.05 m depth.

Channel complexity

Parameters drawn to capture channel complexity were the sinuosity, the variance in river width and depth and the average sediment sorting-coefficient (S_o). Ratios of the parameter values of the selected reaches were calculated and subsequently summarized for estimation of channel complexities.

Physico-chemical water analysis

Within the scope of the SAE and the monitoring project (see table 1; sampling and analysis were realized by the monitoring team) water samples for solute chemistry and particle concentrations analysis were taken on a weekly to monthly basis during the vegetation period of 2002. Sampling sites included the SAE reaches, all main hydrologic in- and outputs of the entire FCR-chain as well as the tributaries. Samples were taken approximately 0.2 m below the water surface with PE-HD bottles (pre-rinsed with deionized-water), were stored in a refrigerator box and on the same day of sampling prepared for subsequent analyses.

In the field water temperature (WTW OXI 330/CellOx 325), dissolved oxygen (WTW OXI 330/CellOx 325), specific conductivity (WTW LF 330/TetraCon 325) and pH (WTW PH 330/SenTix 21) were measured. In addition water temperature was recorded in 30 – 60 minute intervals with single-point data-loggers (Diligence^{TM}, Comark Ltd.) exposed at the head of FCR 3, 6 and 7.

In the laboratory alkalinity was defined through potentiometric titration with 0.1 M HCl (down to a pH of 4.3 with a pH-meter Metrohm 651) according to German Standards (DIN EN ISO 9963-1:1996 and 9963-2:1996). Analysis of total phosphorous (P_{total}) and total nitrogen (N_{total}) was performed on unfiltered samples. A subsample was filtered through pre-ashed glass-fiber filters (47 mm GF/F_{fine}, 2.5 h at 490 °C). Total suspended solids (TSS) and particulate organic matter (POM) were determined using weighted and combusted GF/F_{fine} from previous subsample filtration. The filtrate was used for analysis of dissolved P (P_{sol}) and N (N_{sol}) fractions and dissolved organic carbon (DOC). DOC concentrations were measured with a Shimadzu TOC-5000 analyzer. Aliquots to quantify P_{total} and P_{sol} as well as N_{total} and N_{sol} (modified from Kjeldahl 1883) were digested with sulphuric-acid and peroxide before colorimetric analysis. Particulate P (P_{part}) and N (N_{part}) were obtained through subtraction of soluble parts from the total. Dissolved organic P (DOP) and N (DON) were obtained through subtraction of phosphate (P-PO_4) and ammonium (N-NH_4) respectively from the soluble fraction. P_{total}, P_{sol}, soluble reactive phosphorus (SRP/P-PO_4), N-NH_4, nitrite (N-NO_2), nitrate (N-NO_3) and chloride (Cl) concentrations were measured colorimetrically as following: P as reduced alpha-P-molybdenum-acid (λ 890 nm; modified from DIN EN 1189:1996), N-NH_4 as indophenol-blue sequent Berthelot-reaction (λ 700 nm; modified from Solórzano 1969 and Austrian Standard OENORM ISO 7150-1:1987), N-NO_2 as azo-dye (λ 542 nm; DIN EN 26777:1993), N-NO_3 as nitro-salicylate (λ 412 nm; DIN 38405-29:1994) and Cl as Fe-III-thio-cyanate (λ 490 nm; modified from Küffer *et al.* 1975).

Subsamples for chl-*a* measurements were filtered through glass-fiber filters (47 mm GF/C_{coarse}). Filters were homogenized and treated with ultrasound in 90 % acetone. For the further chl-*a* extraction and calculation procedure see sub-chapter ‚Sediment composition' and eq. 5.

Solute addition experiment

The objective of the SAE was to quantify the transient-storage and nutrient-retention potential of two differently restored river segments. Therefore the experiment took place during low flow conditions to receive data of the river in processing mode. The SAE comprised three single solute additions (July/August/September 2002) to the respective reaches. It was crucial that the reach length was greater than the length of the calculated enriched water-parcel (eq. 4). The conservative tracer NaCl and the reactive nutrients N (as NH_4Cl) and P (as NaH_2PO_4) were injected simultaneously. NaCl served for estimating total transient-storage. This approach was recommended at the Stream Solute Workshop (1990) for being the more practical to calculate spiraling metrics, rather than the isotopic tracer technique. Yet, as being the rough method, marginal differences in spiraling metrics remains undiscernibly, therefore application was advised for distinctly contrasting reaches (Hanafi *et al.* 2006). Further no differentiation between biological and geochemical nutrient retention was possible. Neither could potential nutrient gains from the hyporeal or riparian zone during the SAE be taken into account. Data was fit to a one-dimensional solute storage-transport model (STREAMES protocols 2001, p. 68; Hart 1995). Inherent part of the model was a mass balance.

The slug-type (very short-term) injection method was chosen since background chloride concentrations were high. A constant-type injection would have asked for excessive amounts of tracer reagent to be added or even exceed solubility in the carboy solution. The carboy solution (16 – 20 L) was released all at once at the reach top. At the reach bottom water samples were taken, at the center of the cross section, according to changes in conductivity records. Sampling intervals were within seconds to minutes depending on how fast the slug passed the monitoring site. During slug movement conductivity is observed rising, peaking and proceeding declining (Allan 1995). Sampling was increased at turning points of slug passage to assure a good database for further calculation of curve area extension. Background samples were taken before slug passage and afterwards, to be able to correct for changes in background concentrations during the enrichment experiment. Conductivity values, sample numbers and the according time were recorded simultaneously. The concentrations of $P\text{-}PO_4$, $N\text{-}NH_4$ and Cl-NaCl were measured colorimetrically in the laboratory (see sub-chapter 'physico-chemical water analysis'). $N\text{-}NO_2$ and $N\text{-}NO_3$ were additionally measured to help interpret nitrogen dynamics.

Estimation of the enriched water-parcels length and volume (V_{parcel}, eq. 5) were required to calculate the solutes concentrations in the carboy solution (C_{carboy}, eq. 6) and to obtain the masses of salts added (STREAMES protocols 2001, p. 64). Data applied for the calculation of water-parcel length and volume was gained from preliminary tracer-experiments. T_1 is the point in time of slug passage at the start of conductivity increase and t_{peak} the conductivity peak-point of time.

$$\text{Water-parcel length (m)} = \text{t}_{\text{peak}} - \text{t}_1\,(\text{s}) * \text{flow velocity}\,(\text{m}\,\text{s}^{-1})$$

eq. 6

$$\text{Water-parcel volume, V}_{\text{parcel}}(\text{L}) = \left(\text{water-parcel length}\left(\text{m}\right) * \text{river width}\left(\text{m}\right) * \text{river depth}\left(\text{m}\right)\right) * 1000$$

eq. 7

$$\text{Carboy concentration, C}_{\text{carboy}}\left(\text{mg L}^{-1}\right) = \frac{\left(\text{C}_{\text{SAE}}\left(\text{mg L}^{-1}\right) * \text{V}_{\text{SAE}}\left(\text{L}\right)\right) - \left(\text{C}_{\text{background}}\left(\text{mg L}^{-1}\right) * \text{V}_{\text{parcel}}\left(\text{L}\right)\right)}{\text{V}_{\text{carboy}}\left(\text{L}\right)}$$

eq. 8

V_{carboy} is the carboy volume added to the reach during the SAE. V_{SAE} is V_{parcel} plus V_{carboy}. $C_{background}$ is the solute concentration at ambient levels. C_{SAE} is the desired solute concentration in the water-parcel after the slug addition.

A one and a half increase of the ambient Cl-NaCl, and the twofold of P-PO_4 and N-NH_4 concentrations was desired to guarantee (i) the detection of the enriched water-parcel in the field, (ii) a clear distinction between the base-line and the slug-curve concentrations and (iii) a sufficient solute supply for the estimate of retention potentials. At the same time it was crucial to consider that biotic uptake processes follow saturation kinetics. It had been observed that an increase in nutrient concentration greater than the twofold can saturate a systems capacity to utilize a nutrient and may lead to a decrease in uptake rate and velocity or elongation of uptake length (Newbold *et al.* 1982; Thomas *et al.* 2003; O'Brien *et al.* 2008). Cl, P-PO_4 and N-NH_4 background estimates had been averaged from River Wien monitoring data of previous years.

Total transient storage

The conductivity records over time were the basis for total transient storage calculations. A sharp increase in conductivity at the initial and a sharp decrease at the end of the slug passage indicates that hydrological retention is small or non-existent. The conductivity-time curve is symmetrical. In contrast, a gradual increase at the rising shoulder and a long tailing at the descending limb of the curve reflect existence of transient storage (Allan 1995). Parameters following (STREAMES protocols 2001, p. 58) were used as descriptors of transient storage in the River Wien system.

- Minimum (v_{min}), average (v_{avg}) and maximum (v_{max}) flow-velocity were calculated as:

$$\text{Minimum flow-velocity, } v_{min}\left(\mathrm{m\,s^{-1}}\right) = \frac{\text{reach length}\left(\mathrm{m}\right)}{\text{time when conductivity is back to ambient level}\left(\mathrm{s}\right)}$$

eq. 9

$$\text{Average flow-velocity, } v_{avg}\left(\mathrm{m\,s^{-1}}\right) = \frac{\text{reach length}\left(\mathrm{m}\right)}{\text{time when conductivity reaches the peak}\left(\mathrm{s}\right)}$$

eq. 10

$$\text{Maximum flow-velocity, } v_{max}\left(\mathrm{m\,s^{-1}}\right) = \frac{\text{reach length}\left(\mathrm{m}\right)}{\text{time when conductivity first increases}\left(\mathrm{s}\right)}$$

eq. 11

- The parameter slug passage duration (*SPD*, min) is a measure of the water turnover-time within an experimental reach. It is the time period between conductivity first increase and return to ambient level.

- The ratio of v_{min} to v_{avg} is an indicator of total transient storage. Values close one suggest negligible hydrological retention. Values distinct smaller one indicate occurence of transient storage at a given section.

Calculations of further transient storage parameters K_1, K_2 and extent of the transient storage-zone (*TSZ*) were derived from the STREAMES protocols (2001, p. 58–61) and Hart (1995). All conductivity data used for computation were corrected for background values. In cases where the background conductivity did not remain constant during the SAE, correction accounted for the ambient variation over time.

- K_1 is the water exchange rate from the free flowing water-column to the *TSZ* (Fig. 4A).

$$K_1\left(\mathrm{s^{-1}}\right) = \left(\frac{\text{average flow-velocity } v_{avg}\left(\mathrm{cm\,s^{-1}}\right)}{\text{reach length}\left(\mathrm{cm}\right)} * -1\right) * \mathrm{Ln}\left(\frac{R\left(\mathrm{\mu S\,cm^{-1}\,s}\right)}{I\left(\mathrm{\mu S\,cm^{-1}\,s}\right)}\right)$$

eq. 12

I is the total area below the conductivity vs. time curve (Fig. 4A).

$$I\left(\mathrm{\mu S\,cm^{-1}\,s}\right) = \sum \frac{\left(\text{conductivity}_{tx} + \text{conductivity}_{tx-1}\left(\mathrm{\mu S\,cm^{-1}}\right)\right) * \left(t_x - t_{x-1}\left(\mathrm{s}\right)\right)}{2}$$

eq. 13

R (µS cm^{-1} s) is the theoretical area below the conductivity vs. time curve if transient storage were non-existent. Data considered for the areal calculation started at the rising point and ended at the peak of the conductivity increase. Subsequent the calculated area was multiplied by two. The area R is symmetrical (Fig. 4A).

Transient storage occurs when the area I is greater than the area R.

- K_2 (s^{-1}) is the water exchange rate from the *TSZ* to the free flowing water-column. K_2 refers to the inclination of the tail of the conductivity vs. time curve (framed blue in fig. 4A). Data considered started at the point in time when the slope between successive conductivity vs. time readings was smallest (circled red in fig. 4B). In cases where there was more than one value being lowest the value corresponding to the longest time was selected. Data for the regression was logarithmized.

- The extent of the *TSZ*, the compartments delaying the longitudinal water downstream transport, was calculated as follows (Fig. 4A):

$$\text{Transient storage-zone, } TSZ\,(\%) = \left(\frac{\text{total area } I\left(\mu\text{S}\,\text{cm}^{-1}\,\text{s}\right) - \text{symmetrical area } R\left(\mu\text{S}\,\text{cm}^{-1}\,\text{s}\right)}{\text{total area } I\left(\mu\text{S}\,\text{cm}^{-1}\,\text{s}\right)} \right) * 100$$

eq. 14

The following parameters derived from K_1 or K_2:

- K_1/K_2 (relative transient storage) is equivalent to the estimate A_S/A – the ratio between the cross-sectional area of the free flowing water-column and the cross-sectional area of the transient storage-zone. The bigger the value of the ratio, the more retention occurs. A_S/A in particular is frequently published and used for result comparison within literature. It is noted that multiple calculation modes for A_S/A are documented.

- T_W (s), inverse of K_1, equals the average time that a water molecule travels in the free flowing water-column before it enters the *TSZ*.

- T_S (s), inverse of K_2, equals the average time that a water molecule remains in the *TSZ* before it re-enters the free flowing water-column.

- The water uptake-length S_{H2O} (m) is the average distance that a water molecule travels in the water column before being transiently retained. S_{H2O} was calculated by multiplying T_W with v_{avg}.

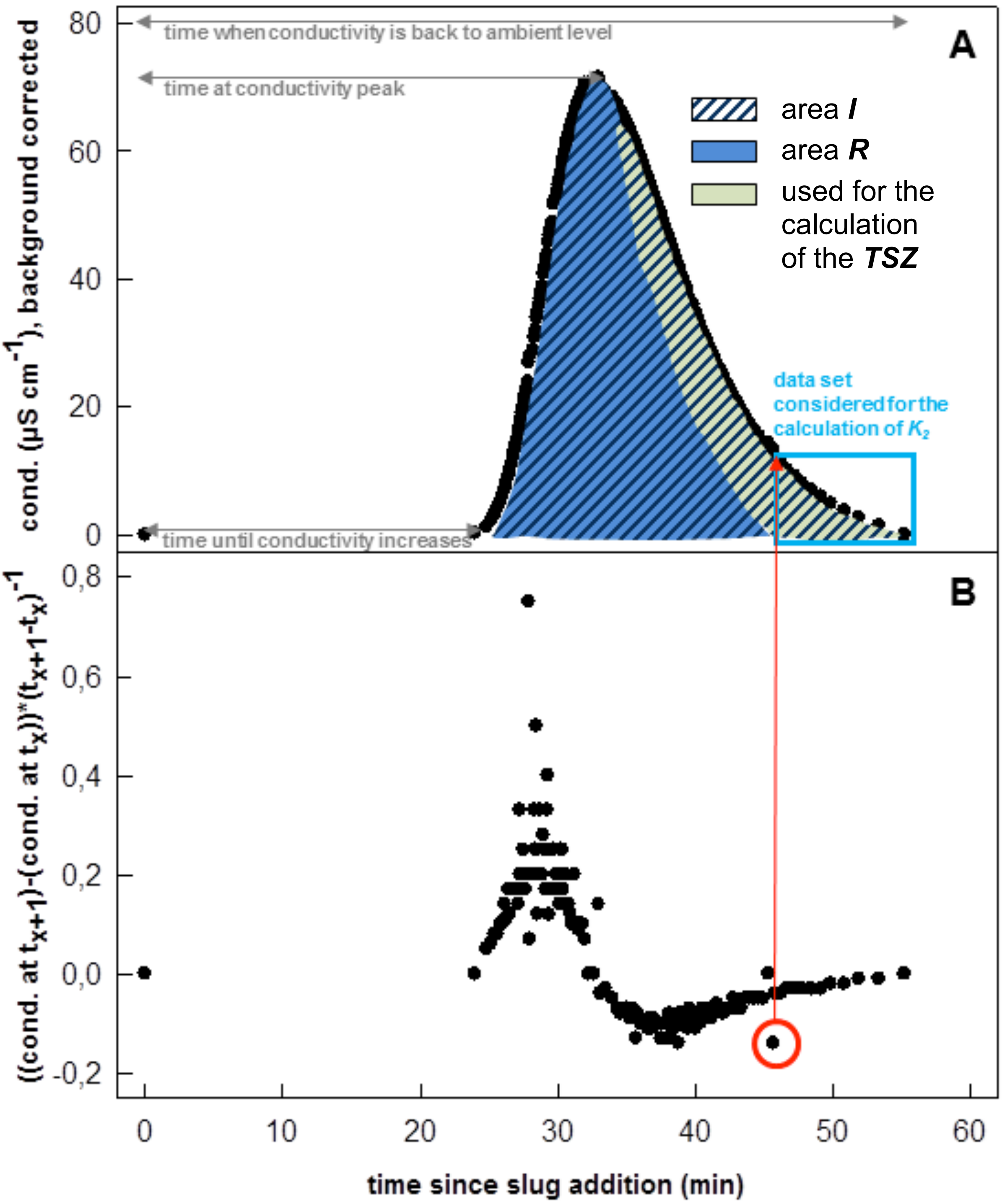


Figure 4 **A**: Course of conductivity (cond., µS cm^{-1}) during time since slug addition (min). Conductivity readings were corrected for the background value. The data set considered for the calculation of K_2 (s^{-1}), the water exchange-rate from the transient storage-zone to the free flowing water-column, is framed turquoise. The total area below the conductivity vs. time curve considered for the calculation of *I* (µS cm^{-1} s) is dark-blue striped. The theoretical area below the conductivity vs. time curve if transient storage were non-existent, considered for the calculation of *R* (µS cm^{-1} s), is royal blue colored. The area below the conductivity vs. time curve considered for the calculation of the transient storage-zone (*TSZ*, %) is colored olive-green. **B**: Slope values of consecutive conductivity readings per time since slug addition. The red-circled value indicates the starting point of the regression for the calculation of K_2 (s^{-1}). Data for figure 4 derived from the SAE on the 28th of July 2002 in reach II.

Nutrient spiralling

Nutrient spiraling-metrics described below were based upon calculation suggestions made at the Stream Solute Workshop (1990) or derived from the STREAMES protocols (2001, p. 56 and 68) and Butturini & Sabater (1999). Solute values were background corrected. Drifting of solute base lines lead to distortion of results. In cases where the background solute concentrations did not remain constant during the SAE, correction accounted for the ambient variation over time.

To estimate the uptake rate *U* – the parameter all further spiralling metrics derive from – the amount of nutrient retained (mg) along the reach was required.

- First it was necessary to calculate theoretical nutrient concentrations pretending a state of non-geochemical sorption and non-biological uptake – as if no nutrient spiraling (‘no-sp.’) occurred during the slug passage. Transient storage, advection and dispersion were taken into account through calculating the ‘no-sp.’ nutrient concentrations (nut. conc.) with the observed chloride concentrations (‘obs.’ Cl conc.). The observed chloride concentrations accord to the actually measured concentrations.

$$\text{'no-sp.' nut. conc.}\left(\text{mg}\,\text{L}^{-1}\right)=\frac{\text{'obs.' Cl conc.}\left(\text{mg}\,\text{L}^{-1}\right)}{\text{Cl carboy conc.}\left(\text{mg}\,\text{L}^{-1}\right)}*\text{nutrient carboy conc.}\left(\text{mg}\,\text{L}^{-1}\right)$$

eq. 15

- Second the ‘no-sp.’ and ‘obs.’ total areas below the nutrient vs. time curve had to be calculated. The observed nutrient concentrations accord to the actually measured concentrations.

$$\text{'no-sp.' total area}\left(\text{mg}\,\text{L}^{-1}\,\text{min}\right)=\sum\frac{\left(\text{'no-sp.' nut. conc.}_{tx}+\text{'no-sp.' nut. conc.}_{tx-1}\left(\text{mg}\,\text{L}^{-1}\right)\right)*\left(t_x-t_{x-1}\left(\text{min}\right)\right)}{2}$$

eq. 16

$$\text{'obs.' total area}\left(\text{mg}\,\text{L}^{-1}\,\text{min}\right)=\sum\frac{\left(\text{'obs.' nut.conc.}_{tx}+\text{'obs.' nut.conc.}_{tx-1}\left(\text{mg}\,\text{L}^{-1}\right)\right)*\left(t_x-t_{x-1}\left(\text{min}\right)\right)}{2}$$

eq. 17

- The difference of the ‚obs’ and ‚no-sp.’ total areas was considered for the calculation of the amount of nutrient retained (Fig. 5, green colored area). Geochemical and biological nutrient retention was referred to as occurring when the ‘obs.’ total area was smaller than the ‘no.-sp.’ total area below the nutrient vs. time curve.

$$\text{nutrient retained}\left(\text{mg}\right)=\left(\text{'no-sp.' total area}-\text{'obs.' total area}\left(\text{mg}\,\text{L}^{-1}\,\text{min}\right)\right)*\text{Q}\left(\text{L}\,\text{min}^{-1}\right)$$

eq. 18

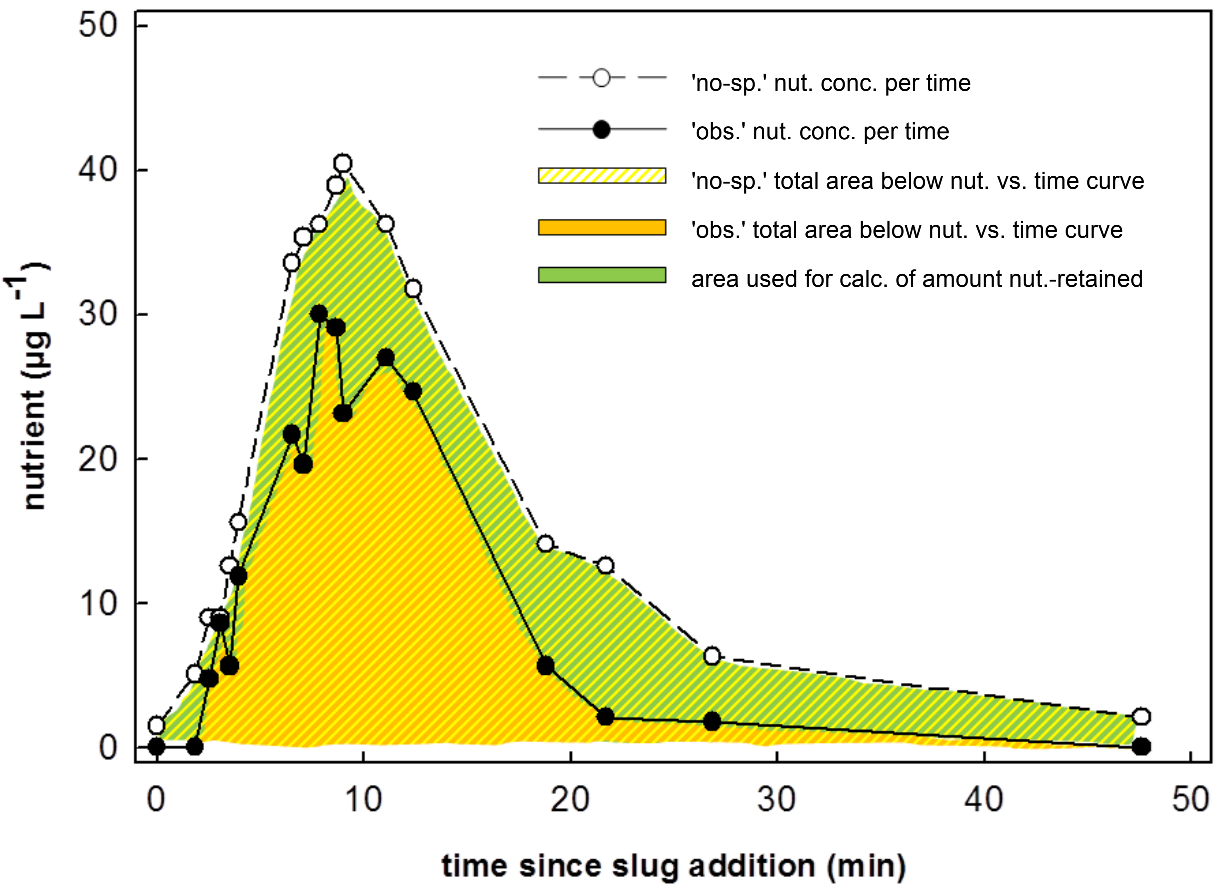


Figure 5 Course of the no-spiraling (‘no-sp.’, dashed line, empty symbol) and the observed nutrient concentrations (‘obs.’ nut. conc., solid line, full symbol) during slug passage. The ‘no-sp.’ total area below the nutrient vs. time curve (µg L^{-1} min) is indicated through the yellow striped area. The ‘obs.’ total area below the nutrient vs. time curve is indicated through the orange colored area. The area used for the calculation of the amount of nutrient retained is indicated through the green colored area. Data for fig. 5 derived from P-PO_4 values of the SAE on the 28th of July 2002 at reach I.

The uptake rate *U* is a measure of dissolved nutrient mass withdrawn from the water column per unit area and time (Newbold *et al*. 1982). *U* is referred to as the capacity of nutrient processing and retention (Newbold 1992). It is the most cogent and common metric for reporting nutrient uptake. The spiralling metrics deriving from *U* are the uptake length S_W, uptake-rate coefficient K_C and mass-transfer coefficient V_f.

The **uptake rate at experimental level *U_{exp}*** is an indicator for the potential (or even maximum) adsorption and metabolic utilization of nutrients. The addition time (min) is the time span from the injection of the carboy until the conductivity returned to background level.

$$\text{Uptake rate at experimental level, } U_{exp}\,(\text{mg m}^{-2}\,\text{min}^{-1}) = \frac{\text{nutrient retained}(\text{mg})}{\text{stream bottom area}(\text{m}^2) * \text{addition time}(\text{min})}$$

eq. 19

The **uptake length *S_W*** is defined as the average downstream travel distance of a nutrient molecule as inorganic solute in the water column prior to removal (Newbold *et al*. 1981). It is a theoretical metric of average uptake length, not measured but calculated. In general S_W is longer than most of the study reaches. Nevertheless positive values are obtained for *U* as nutrient uptake occurs within a reach. S_W is an indicator of nutrient utilization efficiency relative to supply available (Newbold *et al*. 1981).

$$\text{Uptake length, } S_W\,(\text{m}) = \frac{\text{peak nutrient conc.}(\text{mg L}^{-1}) * \text{Q}(\text{L min}^{-1})}{U_{exp}(\text{mg m}^{-2}\,\text{min}^{-1}) * \text{wetted perimeter}(\text{m})}$$

eq. 20

The turnover length S_B equals the nutrient travel distance within the biota, in organic particulate form, until regeneration (Allan 1995; Newbold 1992). Estimation of S_B requires use of isotopic tracers (Ashkenas *et al*. 2004). S_W and S_B sum up to the spiraling length *S*, the stretch "over which the downstream flux is cycled, on average, once" (Newbold 1992). S_B is known to be short, the minor part of the spiraling length. A river section is referred to as being highly efficient when spirals are tight; hence the intensity of nutrient-flux reutilization is greater than transport (Webster 1975, Webster & Patten 1979, Newbold *et al*. 1982). The continual "nutrient transport is retarded relative to that of water" (Newbold 1992).

The **uptake rate at ambient level *U_{amb}*** is a measure for estimation of nutrient uptake during ambient conditions. Results of U_{amb} were used for comparison with results of mass balancing.

$$\text{Uptake rate at ambient level, } U_{amb}\,(\text{mg m}^{-2}\,\text{min}^{-1}) = \frac{\text{stream nutrient conc.}(\text{mg L}^{-1}) * \text{Q}(\text{L min}^{-1})}{\text{uptake length } S_w\,(\text{m}) * \text{wetted perimeter}(\text{m})}$$

eq. 21

The metric **uptake rate coefficient *K_C*** is corrected for flow-velocity and used for calculation of the mass transfer coefficient V_f. Therefore both K_C and V_f are appropriate for comparison of nutrient spiraling between streams differing in hydrological conditions (STREAMES protocols 2001, p. 54).

$$\text{Uptake rate coefficient, } K_c\,(1\,\mathrm{s}^{-1}) = \frac{\text{average flow-velocity } v_{avg}\left(\mathrm{m\,s}^{-1}\right)}{\text{uptake length } S_w\left(\mathrm{m}\right)}$$

eq. 22

The **mass transfer coefficient *V_f*** is a measure for how rapidly a nutrient molecule is retained, or cycling occurs (Meyer *et al.* 2005) and an estimate of nutrient demand relative to supply (Hall *et al.* 2002). V_f is also referred to as uptake velocity (Bukaveckas 2007).

$$\text{Mass transfer coefficient, } V_f\,(\mathrm{m\,s}^{-1}) = \text{depth}\left(\mathrm{m}\right) * \text{uptake rate coefficient } K_c\left(1\,\mathrm{s}^{-1}\right)$$

eq. 23

Mass balance

Mass balances were calculated of TSS, P_{total} and N_{total}, the particulates P_{part} and N_{part} and the solutes $P\text{-}PO_4$, $N\text{-}NH_4$, $N\text{-}NO_3$, DOP, DON and DOC. Estimations were realized with data of reaches I – III (04/2002 – 10/2002). Dates considered for mass balancing showed water temperatures above 10°C. Results during low-, mean flow and high water were analyzed separately. Mass balancing was computed as:

$$\text{Residual mass, M}\left(\mathrm{kgm}^{-2}\mathrm{d}^{-1}\right) = \frac{\left(\mathrm{Q_{in}}\left(\mathrm{Ld}^{-1}\right) * \mathrm{C_{in}}\left(\mathrm{kgL}^{-1}\right)\right) - \left(\mathrm{Q_{out}}\left(\mathrm{Ld}^{-1}\right) * \mathrm{C_{out}}\left(\mathrm{kgL}^{-1}\right)\right)}{\text{wetted surface}\left(\mathrm{m}^2\right)}$$

eq. 24

M is the residual mass and was extrapolated to the wetted surface. Positive residuals indicate a sink of matter, negative a source. Q_i is the input discharge and C_i the input concentration at the top of a given river segment. Q_o and C_o are the output discharge and concentration at the bottom of a given river segment. Hager (2002, p. 11) found no significant difference between in- and output discharges at the FCRs. The hydrological balance of the FCRs was assumed to be at steady state. In the present study at dates where either only Q_i or Q_o had been measured from the municipal water authority in- and outflow discharge for calculating MB therefore was set even. At monitoring dates where neither Q_i nor Q_o was available discharge of the permanent gauge Pegel-Hadersdorf was taken.

Statistics

Statistical analysis such as two-parameter correlations was realized with Excel 2011.

RESULTS

Hydrology

After the extremly dry years 2000 – 2001 ($0.43 \pm 0.49\ m^3\ s^{-1}$, n = 720) the mean discharge doubled in 2002 ($0.87 \pm 1.13\ m^3\ s^{-1}$, n = 242). The hydrograph showed several high flash floods. During the vegetation period from April to October floods occurred in June ($4.61\ m^3\ s^{-1}$), August ($8.85\ m^3\ s^{-1}$) and October ($6.85\ m^3\ s^{-1}$; Fig. 6).

During the SAE discharge was very alike at reach I and II. Average values were $0.32 \pm 0.15\ m^3\ s^{-1}$ (n = 3) and $0.34 \pm 0.09\ m^3\ s^{-1}$ (n = 3) respectively (Tab. 5). The confluence with the stream Mauerbach increased the discharge at reach III by 33 % ($0.55 \pm 0.13\ m^3\ s^{-1}$, n = 3).

The brook Rotwassergrabenbach and Grünauerbach, both carrying water only after heavy rainfalls and contributing in general with one to five percent to the River Wien discharge, were dry during the SAE.

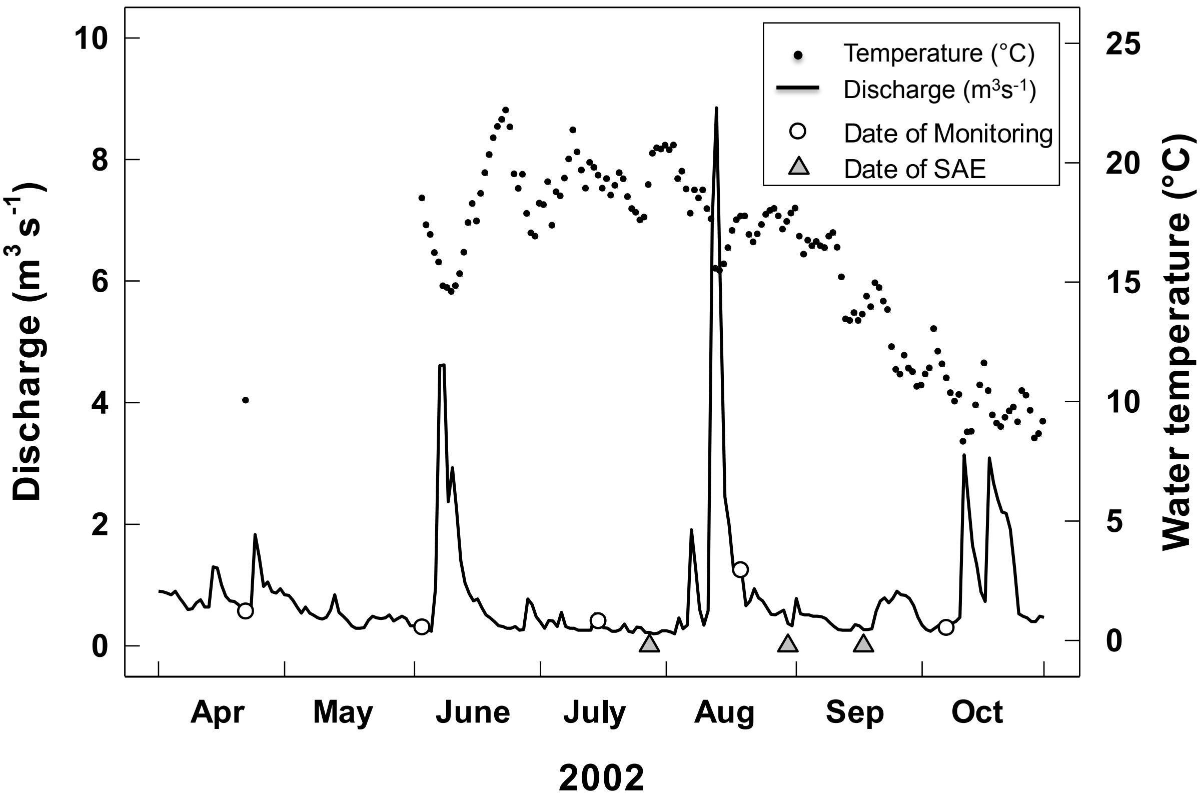


Figure 6 Daily mean discharge (bold line; n = 242; permanent measuring gauge 200 m upstream of the FCRs) and water temperature (dotted line; n = 153; data logger at the head of FCR 6) at the River Wien System from April – October 2002. White circles show the monitoring dates of the physico-chemical sampling. Grey triangles show the Solute Addition Experiment (SAE) study dates.

Riverbed structure

Channel morphology

As a result of particular rehabilitation measures differences in channel morphology (i.e., sinuosity and variance of river width and depth; Tab. 2) were found between the restructured Reaches I and II. Sinuosity was largest and water width and depth (m) CV of transects were highest in the reactivated wetland section. Maximum width even exceded 17 m at the widest measured transect. CV of river widths exceeded reach I by the four times. Reach I and III, in comparison, although exhibiting similar CV widths, showed distinct differences in CV of depths. Reach I exceeded reach III by nearly the three times.

Table 2 Descriptive geomorphologic parameters of study reach I – III. Presented are the values from singular data collections of reach- and shoreline-length. The mean, minimum (Min), maximum (Max) and coefficient of variation (CV) are shown for measured values of river width and depth of reach transects. Measurements took place during low water, with exception of dGPS-surveying of the shoreline length of reach II realized at mean-water level.

		Reach I revitalized riffle-pool					**Reach II reactivated wetland**					**Reach III channelized run**				
Parameter	**Unit**	**Value Mean**	**Min**	**Max**	**CV**	**n**	**Value Mean**	**Min**	**Max**	**CV**	**n**	**Value Mean**	**Min**	**Max**	**CV**	**n**
Reach length	**(m)**	400					280					540				
Shoreline length	**(m)**	491					368					557				
Sinuosity	**/**	1.23					1.32					1.03				
River width	**(m)**	5.80	4.25	8.00	0.18	21	5.92	2.25	17.75	0.87	15	6.66	4.10	8.55	0.13	26
River depth	**(m)**	0.22	0.12	0.37	0.34	491	0.31	0.11	0.60	0.49	359	0.15	0.12	0.21	0.12	722

Sediment composition

Regarding the River Wien sediment composition recorded grain sizes were: silt < 0.06 mm (pelal), sand 0.06 – 2 mm (psammal), granule 2 – 6.3 mm (akal, *fine*), small pebbles 6.3 – 20 mm (akal, *coarse*), pebbles 20 – 63 mm (microlithal), cobbles 63 – 200 mm (mesolithal) and large cobbles > 200 mm (macrolithal).

Detailed SAE study sections sediment composition descriptive data are visualized in figure 7 and listed in table 3.

Grain sizes at reach I ranged from silt to cobbles. Sediments showed a well-mixed distribution, reflected in the highest value for the mean sorting coefficient (S_o) despite the silt and sand fraction accounting for less than 10 %. Differing needs for restoration exhibted a 21 % higher areal coverage with cobbles > 200 mm in comparison to the reactivated wetland.

Substratum at reach II was comprised to 59 % by pebbles and cobbles (20 – 200 mm), to 23 % by silt and sand and to 17 % by granule and small pebbles. Average estimate of Q_{75} (63 mm) and the smallest value for skewness, $S_{k,}$ reflect the dominance of small fractions. Q_{75} for both other study sections was 2.4 and 5 times higher. Further the mean organic matter content (AFDW) of surface sediments exceeded 2 kg per square meter within the reactivated wetland. The revitalized riffle-pool held 2.5 and the channelized run 3.5 times less organic material.

Large cobbles covered more than 80 % of reach III riverbed. Found fractions of pebbles and granule derived from upstream sections and rested on the flagstones. Dense filamentous algae growth resulted in more than 10 times higher stocks of chl-*a* (n = 29) per square meter in contrast to the restructured reaches.

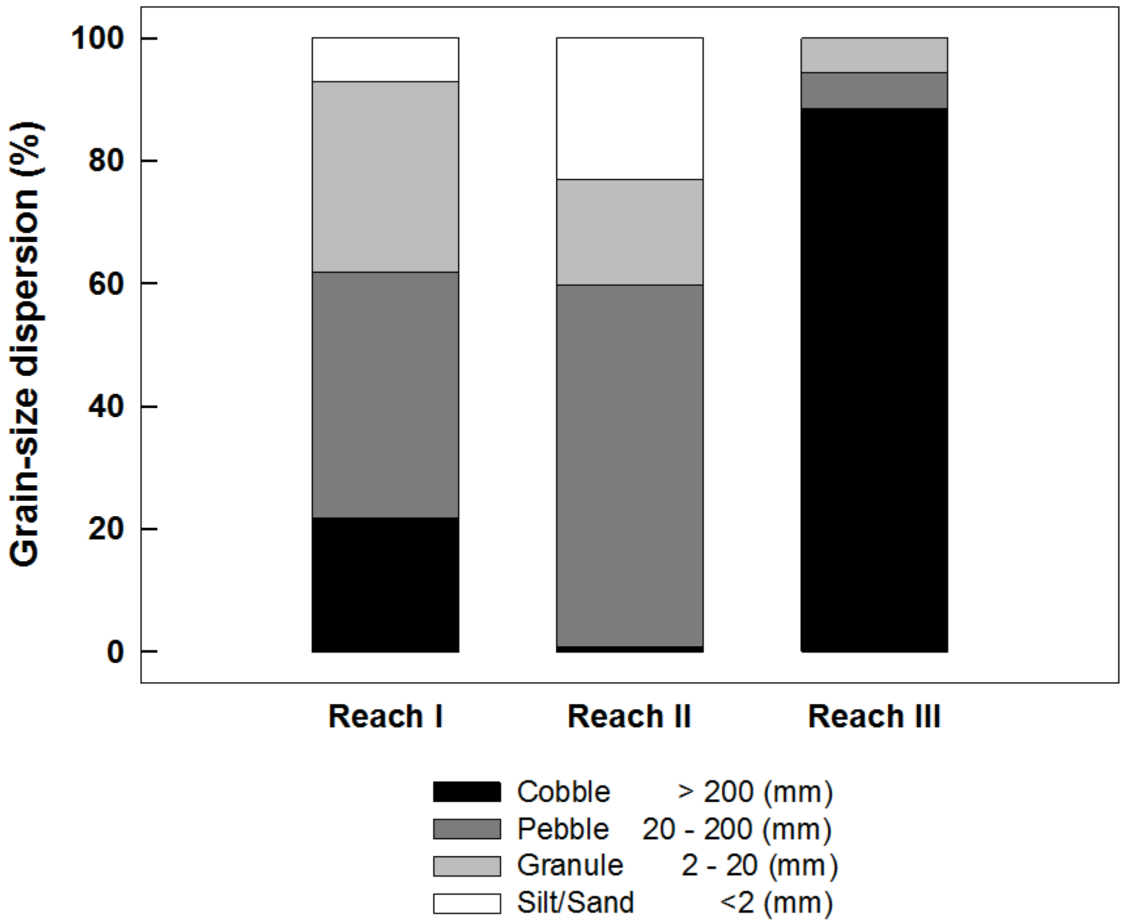


Figure 7 Riverbed sediment grain-size dispersion in percent of the Solute Addition Experiment study reaches I – III.

Table 3 Visually estimated sediment composition and weighted average standing stocks of organic matter (AFDW) and chlorophyll-a (Chl-*a*) at the Solute Addition Experiment (SAE) study reach scale. Listed are the percentiles (Q 25/50/75 % in mm), the sorting (S_o) and the skewness coefficient (S_k). AFDW and chl-*a* calculations refer to 1 m² stream-bottom area with a profundity of 0.05 m. SD indicates the standard deviation.

Parameter	Unit	Reach I revitalized riffle-pool			Reach II reactivated wetland			Reach III channelized run		
		Mean		SD	Mean		SD	Mean		SD
Q 25 %	**(mm)**	14	±	13	16	±	15	223	±	75
Q 50 %	**(mm)**	57	±	59	37	±	28	269	±	87
Q 75 %	**(mm)**	152	±	86	63	±	39	316	±	98
S_o	**/**	3.56	±	1.09	3.11	±	2.62	1.28	±	0.28
S_k	**/**	-0.08	±	0.31	-0.16	±	0.21	-0.02	±	0.02
AFDW	**(g m²)**	856	±	135	2131	±	1678	601	±	86
Chl-*a*	**(g m²)**	0.14	±	0.02	0.18	±	0.14	18.31	±	5.95

Channel complexity

The calculation of channel complexity (see methodology) is based on information provided in tables 2 and 3.

The restructured reaches displayed distinct differences in estimated channel complexity. River Wien incorporation in the FCRs particularly brought an increase in variation of river width and depth and stocks of organic debris. The reactivated wetland was marked clearly the more heterogeneous section. Calculated channel complexity was more than the double (2.47) relative to the revitalized riffle-pool section. Improvements in riverbed conditions at the revitalized riffle-pool section, previously heavily modified, enhanced channel complexity by the twofold in comparison to the channelized run. The channelized run obviously exhibited the most monotonous structural riverbed and habitat conditions. This section is geophysical as biological extensively degraded. The gradient of channel complexity ratios between all three reaches was 1.92 : 4.11 : 1 (reach I : II : III).

Physico-hydrochemistry

Looking at the descriptive water physic and chemistry data (Tab. 4; Results were provided by courtesy of the monitoring team) mentionable differences in the following parameters could be seen between the two restructured reaches I and II: From reach I to II conductivity increased from average 584 to 644 µS cm^{-1}. Also the chloride course showed a moderate rise, to the extent of 8 %. The analyzed dissolved inorganic nutrient as well as the dissolved organic phosphorous (DOP), nitrogen (DON) and carbon (DOC) loads showed a steady downstream decrease from the revitalized riffle-pool to the reactivated wetland section. Average concentration of P-PO_4 exhibited a decline of one tenth and N-NH_4 of three quarters. Charge of DOP, DON and DOC decreased by five to six percent. On the contrary mass flow of P and N particulate matter (P_{part}, N_{part}) as well as of total suspended solids (TSS) showed a distinct rise from reach I to II. P_{part} increased by more than one quarter, N_{part} by one fifth and TSS by one third. 30 % of suspended solids were of organic origin. As the stream Mauerbach in general carries lower loads of salts and nutrients, reach III concentrations were minimized.

Table 4 Physico-chemical descriptive parameters of reach I – III in the River Wien. Average data were obtained from April – October 2002. Data were collected on a weekly to monthly basis. Listed are the water temperature, oxygen, field pH, alkalinity, conductivity, chloride (Cl), phospate (P-PO_4), nitrate (N-NO_3), nitrite (N-NO_2), ammonium (N-NH_4), dissolved organic phosphorus (DOP), dissolved organic nitrogen (DON), dissolved organic carbon (DOC), particulate phosphorous (P_{part}), particulate nitrogen (N_{part}), particulate organic matter (POM), relative particulate organic matter (POM_{rel}), particulate inorganic matter (POM) and chlorophyl-*a* (Chl-*a*). SD indicates the standard deviation.

Parameter	Unit	Reach I revitalized riffle-pool				Reach II reactivated wetland				Reach III channelized run			
		Mean		SD	n	Mean		SD	n	Mean		SD	n
Temperature	**(°C)**	15.9	±	4.2	16	13.1	±	5.8	25	16.1	±	3.6	15
Oxygen	**(mg L^{-1})**	10.6	±	1.5	16	10.8	±	2.1	25	12.4	±	2.2	15
Oxygen	**(%)**	110.5	±	13.4	16	104.7	±	13.5	25	128.6	±	22.5	15
pH field	**(-log H^+)**	8.3	±	0.2	16	8.1	±	0.1	25	8.3	±	0.1	15
Alkalinity	**(mmol L^{-1})**	4.4	±	0.8	10	4.4	±	0.5	19	4.6	±	0.3	9
Conductivity	**(µS cm^{-1})**	584.4	±	90.7	16	643.7	±	72.5	25	638.9	±	32.4	15
Cl	**(mg L^{-1})**	33.7	±	6.4	14	36.5	±	6.5	22	31.7	±	4.0	14
P-PO_4	**(µg L^{-1})**	32.1	±	18.7	16	29.0	±	14.8	24	18.6	±	15.0	15
N-NO_3	**(µg L^{-1})**	2166.4	±	331.6	16	2098.0	±	446.4	24	1670.5	±	298.8	15
N-NO_2	**(µg L^{-1})**	36.4	±	20.5	16	21.0	±	10.7	24	13.1	±	7.0	15
N-NH_4	**(µg L^{-1})**	82.4	±	93.6	16	58.5	±	66.7	24	19.3	±	41.4	15
DOP	**(µg L^{-1})**	4.8	±	1.9	10	4.6	±	2.1	19	4.0	±	2.4	9
DON	**(µg L^{-1})**	291.9	±	58.8	10	277.0	±	50.8	19	257.9	±	18.5	9
DOC	**(mg L^{-1})**	4.1	±	1.5	10	3.8	±	1.1	18	3.5	±	0.7	7
P_{part}	**(µg L^{-1})**	49.7	±	59.7	10	63.3	±	51.2	16	30.0	±	26.5	9
N_{part}	**(µg L^{-1})**	125.9	±	124.0	10	150.6	±	124.4	16	82.2	±	68.8	9
POM	**(mg L^{-1})**	3.9	±	3.5	10	5.1	±	3.5	18	3.9	±	2.7	9
POM_{rel}	**(%)**	34.2	±	15.4	10	28.2	±	14.4	18	25.4	±	7.2	9
PIM	**(mg L^{-1})**	14.9	±	21.7	10	23.0	±	24.5	18	14.2	±	13.7	9
Chl-*a*	**(µg L^{-1})**	5.6	±	4.2	10	5.0	±	3.1	18	4.7	±	2.0	9

Solute addition experiment

Total transient storage

Evaluating effects of channel restoration Bukaveckas (2007) experienced transient storage metrics insensitive to discharge (possibly various *TSZs* contributed to flow retardation according to changes in water level). Following this finding and the understanding that ratios of flow-velocities and hydrological exchange rates consider changes in discharge, comparisons of the restructured study reaches with the channelized run were realized. *SPD* and S_{H2O} precautionary were compared only amongst the restructured reaches, similar in discharge.

Figure 8 visualizes nicely differences in slug passage times as inspect on magnitudes of transient storage: The reactivated wetland exhibited slow conductivity increases at the rising shoulder and longest tailing at the descending limb to reach background conductivity. *SPD* was the double compared to the revitalized riffle-pool section (Tab. 5). Further transient storage parameters confirmed these results. Reach II showed the biggest values for *TSZ* (%) and shortest S_{H2O} (m). Theoretical S_{H2O} (m) exceeded reach length, nevertheless transient storage occurred. The water-uptake length calculated for reach II was twice shorter than for reach I.

The restructured sections exhibited noticeable differences amongst v_{min}/v_{avg}, A_S/A and T_W/T_S, but profound distinction was amongst the wetland and the channelized reach indicating highest hydrological retention occurring at reach II. Ratio estimates for reach I took an intermediate position and were in most cases placed closer to those of reach III. Reach III performed poorest concerning total transient storage. Times for T_W (s) over T_S (s) showed that the water remained ~ 2.2 times longer in the *TSZ* of the reactivated wetland and ~ 1.4 times longer in the *TSZ* of the revitalized riffle-pool than in the *TSZ* of the channelized run. T_W in relation to T_S indicated clearly that at reach III the water entered the *TSZ* fast and again left it fast. At reach I and II the river water took more time to enter the *TSZ*, but remained proportionally longer.

Correlations of transient storage metrics with riverbed structure parameters (Fig. 9) are reviewed in the discussion section.

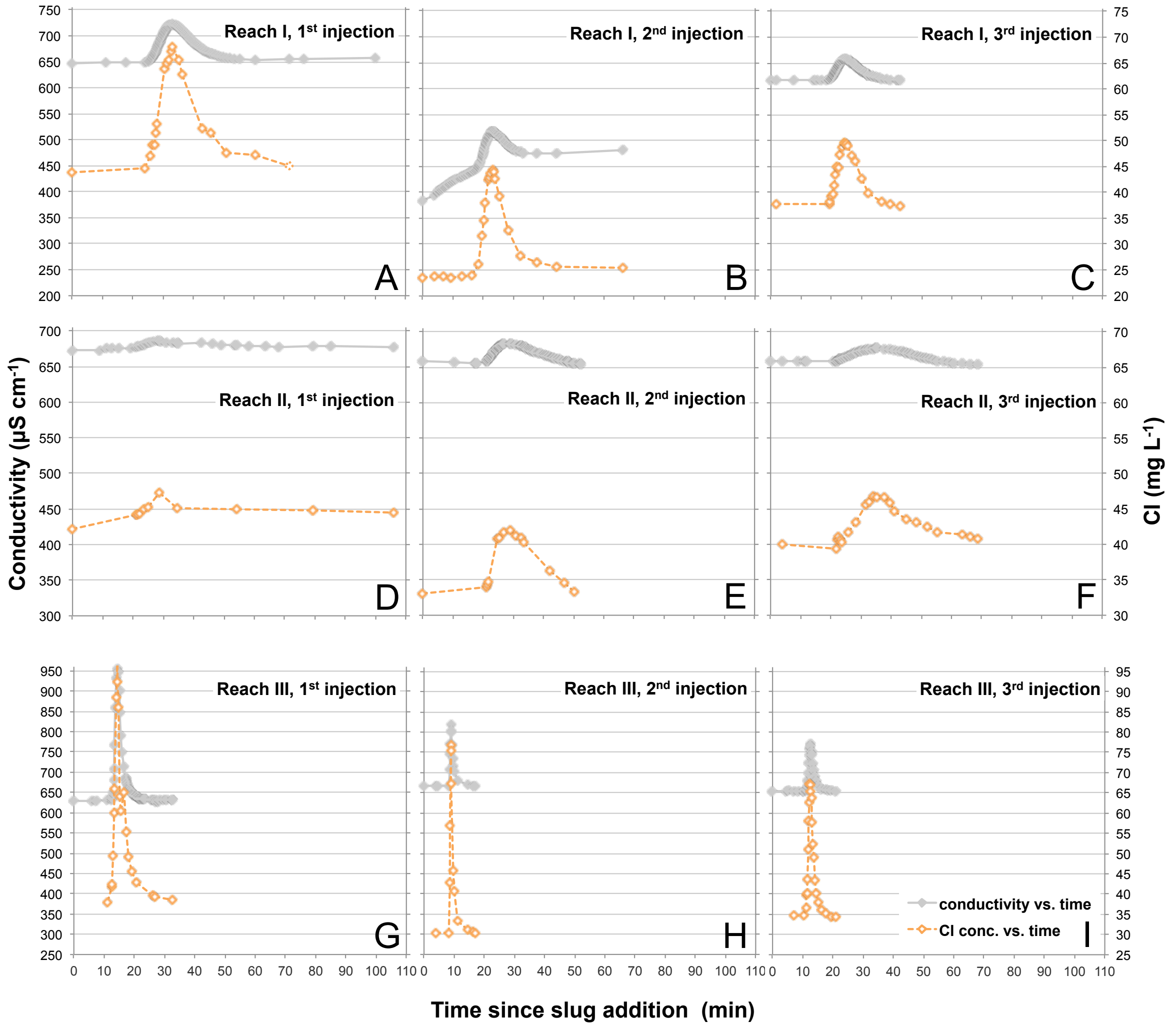


Figure 8 Conductivity run (µS cm^{-1}) and chloride concentration (Cl, mg L^{-1}) curves during slug passage at reach I of the first **A**, second **B** and third **C** solute injection, at reach II of the first **D**, second **E** and third **F** solute injection and at reach III of the first **G**, second **H** and third **I** solute injection. Conductivity data were used for calculating the transient storage parameters. Chloride data served the correction of biogeochemical-nutrient for hydrological retention. Conductivity data are shown as full diamonds and full line, chloride data as empty diamonds and dashed line.

Table 5 Transient storage parameters means and standard deviations (SD) of reach I, II and III. Given are the discharge Q ($m^3\ s^{-1}$), the maximum v_{max}, average v_{avg} and minimum v_{min} water velocity ($m\ s^{-1}$), the ratio of v_{min}/v_{avg}, the slug passage duration *SPD* (min), the water-exchange rate from the free-flowing water to the transient storage-zone K_1 (s^{-1}), the water-exchange rate from the transient storage-zone to the free-flowing water K_2 (s^{-1}), A_S/A as the ratio between the cross-sectional area of the transient storage-zone and cross-sectional area of the free-flowing water, the size of the transient storage-area (*TSZ*, %), the turnover time in the water column T_W and in the transient storage-zone T_S (min) and the water-uptake length S_{H20} (m). For calculating the presented values of Q, v_{max}, v_{avg}, v_{min}, *SPD* and K_2 data from the 3 slug-additions dates was averaged. K_1, A_S/A, *TSZ*, T_W, T_S and S_{H2O} were estimated from the means of the above-mentioned parameters.

Parameter	Unit	Reach I revitalized riffle-pool		Reach II reactivated wetland		Reach III channelized run	
		Mean	SD	Mean	SD	Mean	SD
Q	($m^3\ s^{-1}$)	0.33	0.13	0.34	0.10	0.53	0.12
v_{max}	($m\ s^{-1}$)	0.35	0.08	0.22	0.03	0.86	0.18
v_{avg}	($m\ s^{-1}$)	0.26	0.06	0.16	0,02	0.78	0.20
v_{min}	($m\ s^{-1}$)	0.16	0.04	0.07	0,02	0.51	0.11
v_{min} / v_{avg}		0.59	0.04	0.44	0.15	0.66	0.03
SPD	(min)	25.56	8.62	52.57	28.72	7.32	1.59
K_1	(s^{-1})	$0.03*10^{-2}$	$0.01*10^{-2}$	$0.05*10^{-2}$	$0.03*10^{-2}$	$0.07*10^{-2}$	$0.04*10^{-2}$
K_2	(s^{-1})	$0.45*10^{-2}$	$0.13*10^{-2}$	$0.38*10^{-2}$	$0.38*10^{-2}$	$1.49*10^{-2}$	$0.41*10^{-2}$
A_S/A		0.06	0.02	0.41	0.61	0.04	0.01
TSZ	(%)	33.77	3.53	50.78	20.73	34.70	7.77
T_W	(min)	66.81	22.82	54.13	44.19	30.13	12.88
T_S	(min)	3.85	0.94	11.70	13.28	1,17	0.31
T_W / T_S		17.81	5.59	11.34	9.06	24.87	6.17
S_{H20}	(m)	1006	98	478	322	1314	324

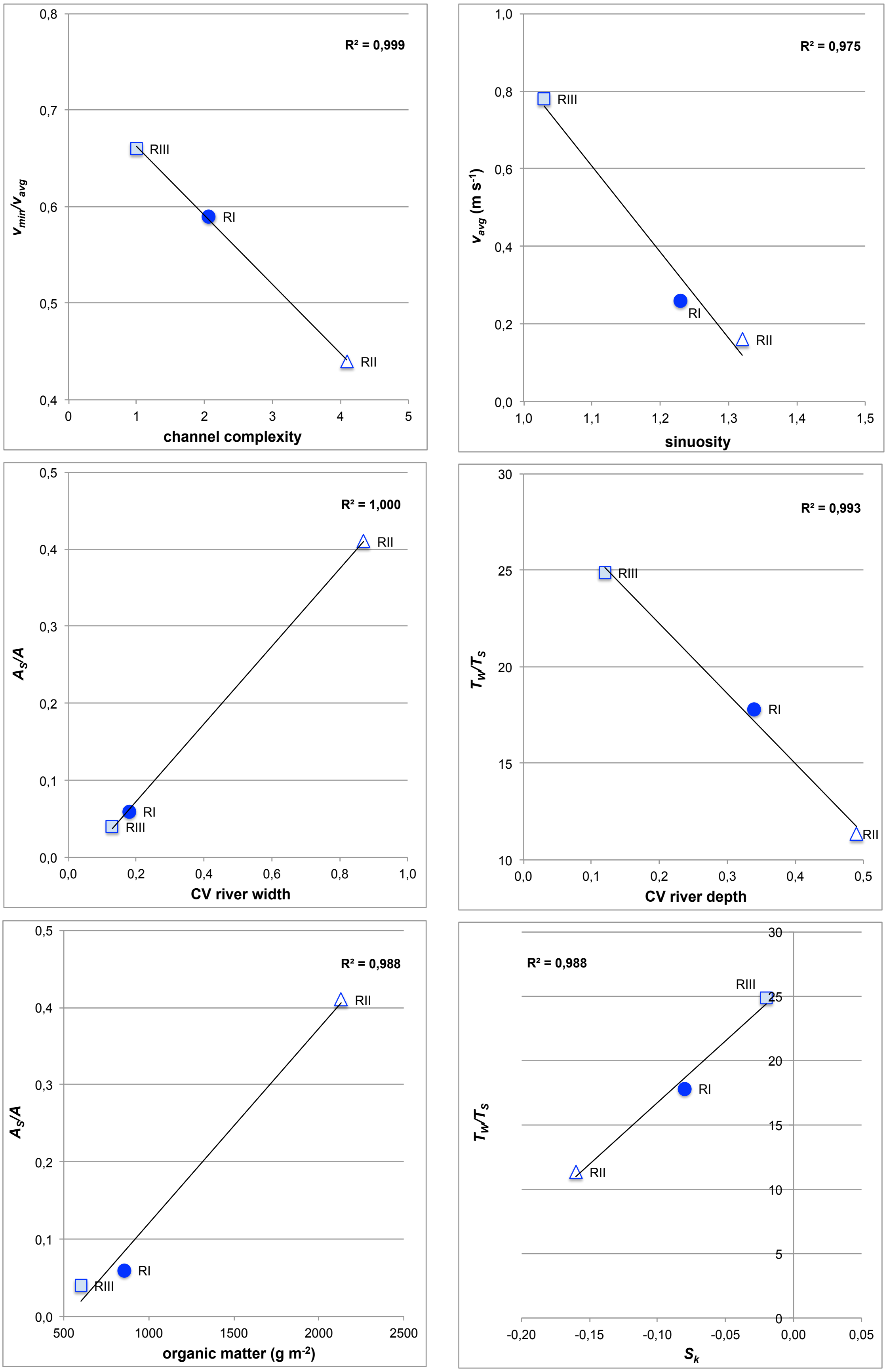


Figure 9 Correlations of transient storage parameters: ratio of minimum to average water velocity v_{min}/v_{avg}, average velocity v_{avg}, ratio between the cross-sectional areas of the transient storage-zone (TSZ) and free-flowing water A_S/A and ratio of the turnover times in the water column and TSZ T_W/T_S with: channel complexity, sinuosity, CV width and depth, organic matter and coefficient of skewness S_k. Data of reach I (RI) is shown as dots, reach II (RII) as triangles and reach III (RIII) as squares.

Nutrient spiraling

Phosphate and ammonium concentrations during slug passage (Fig. 10 and 11) gave evidence on the outcome of the SAE. Optimal slug courses exhibited a Gaussian distribution of both the observed and 'no-spiraling' concentration per time curves. For phosphate this was the case for all injections, with exception of the first at reach II. An ample background increase for both Cl and $P\text{-}PO_4$ was not achieved. Concerning ammonium only one out of three injections per reach offered data usable for further spiraling parameter calculation (Fig. 11 A, E and G). Lack of sufficient background increases (Fig. 11 D and F) and value oscillations of the observed concentration per time curves (Fig. 11 B, C and I) were the cause. Given these losses nutrient spiraling results considered for comparison and further discussion are highlighted bold in table 6. Negative values indicated an export of tracer or nutrient of more than the added. During the SAEs several increases in concentrations of nitrite and nitrate were observed (Fig. 12 B, C, D, G, H and I).

Rough tendencies observed for spiraling of $P\text{-}PO_4$ amongst the restructured reaches indicated that experimental uptake capacities (U_{exp}) and velocities (V_f) were the six- and the threefold higher (Tab. 6) at reach I. Looking at the uptake length (S_W) a $P\text{-}PO_4$ molecule travelled a noticeable shorter distance in the water column in the revitalized riffle-pool (about 2.3 times) than in the reactivated wetland section. Results of the restructured reaches 3rd injection in September, with water temperatures of 15 °C, were very alike. In July the mean water temperature was 22 and in August 18 °C. Distinct differences for the uptake capacities at experimental and ambiental levels were found.

Comparison of the three reaches (Reach I:II:III) at dates of similar discharge exhibited a 3:1:1 ratio for U_{exp}, 22:9:1 for U_{amb}, 0.2:0.4:1 for S_W and 11:3:1 for V_f.

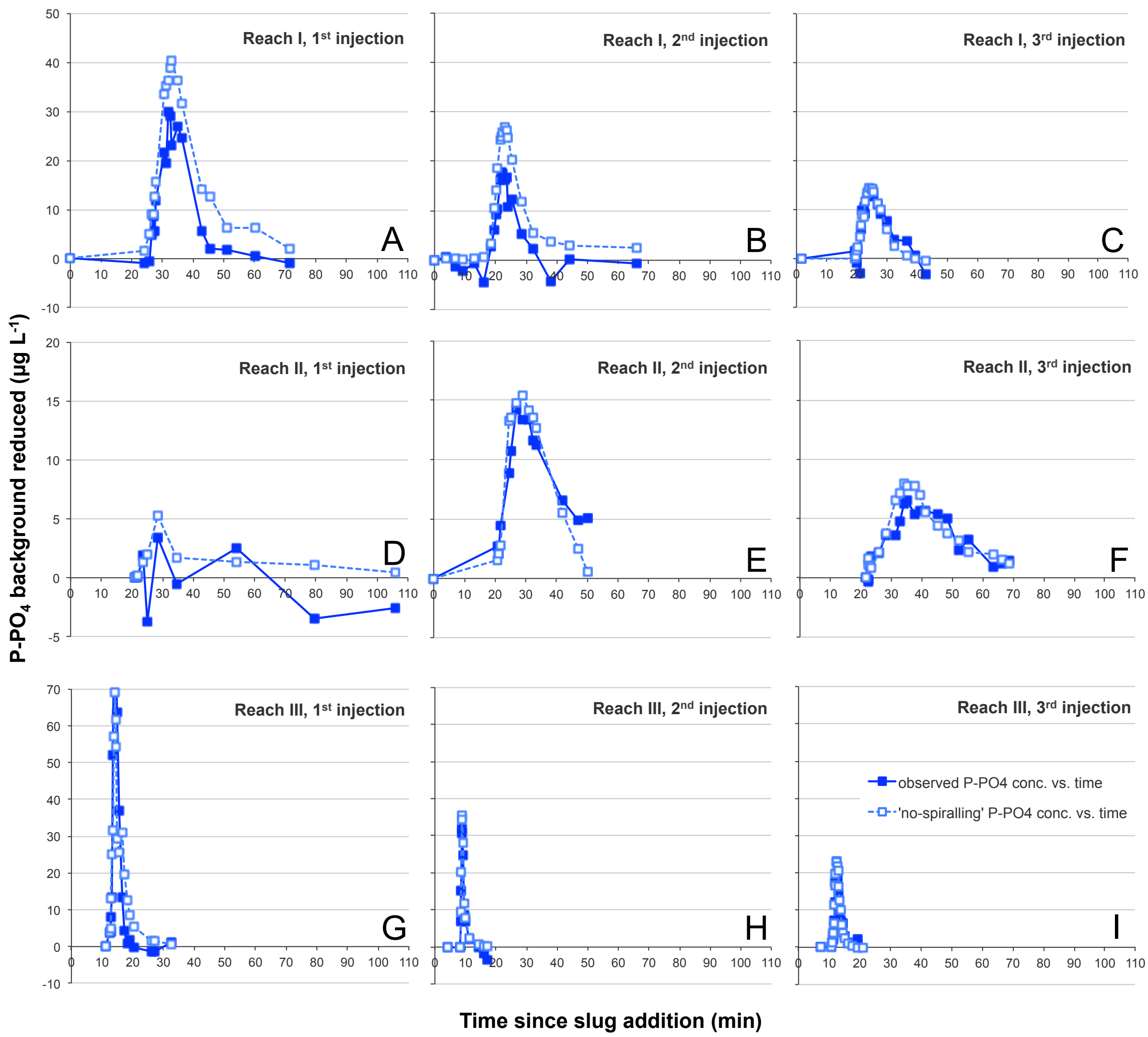


Figure 10 Phosphate concentration (P-PO_4, mg L^{-1}) curves, reduced for background values, during slug passage at reach I of the first **A**, second **B** and third **C** solute injection, at reach II of the first **D**, second **E** and third **F** solute injection and at reach III of the first **G**, second **H** and third **I** solute injection. Data served the calculation of nutrient spiraling parameters. Observed P-PO_4 data is shown as full squares and full line, calculated 'no-spiraling' P-PO_4 data is shown as empty squares and dashed line.

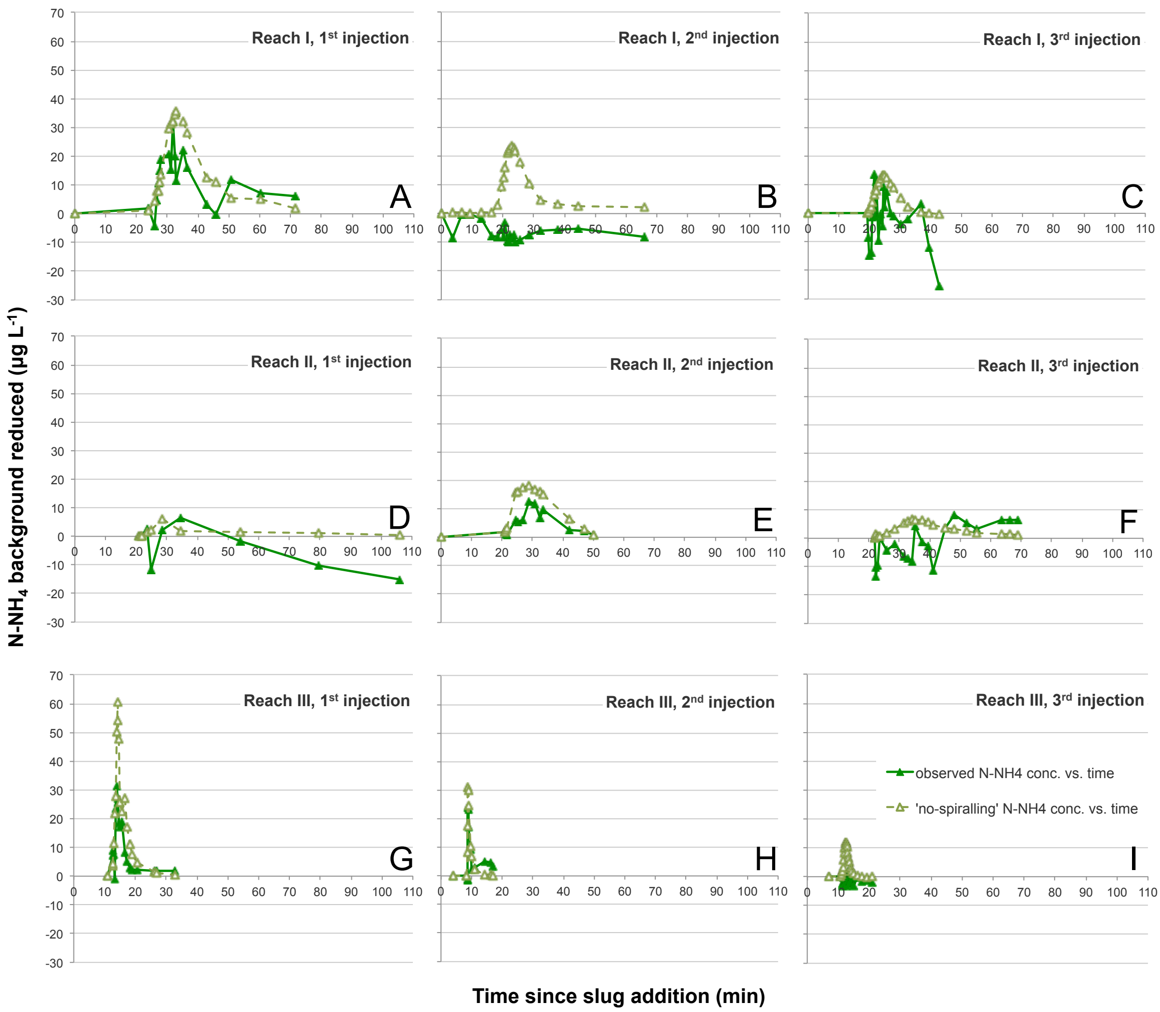


Figure 11 Ammonium concentration (N-NH_4, mg L^{-1}) curves, reduced for background values, during slug passage at reach I of the first **A**, second **B** and third **C** solute injection, at reach II of the first **D**, second **E** and third **F** solute injection and at reach III of the first **G**, second **H** and third **I** solute injection. Data served the calculation of nutrient spiraling parameters. Observed N-NH_4 data is shown as full triangles and full line, calculated 'no-spiraling' N-NH_4 data is shown as empty triangles and dashed line.

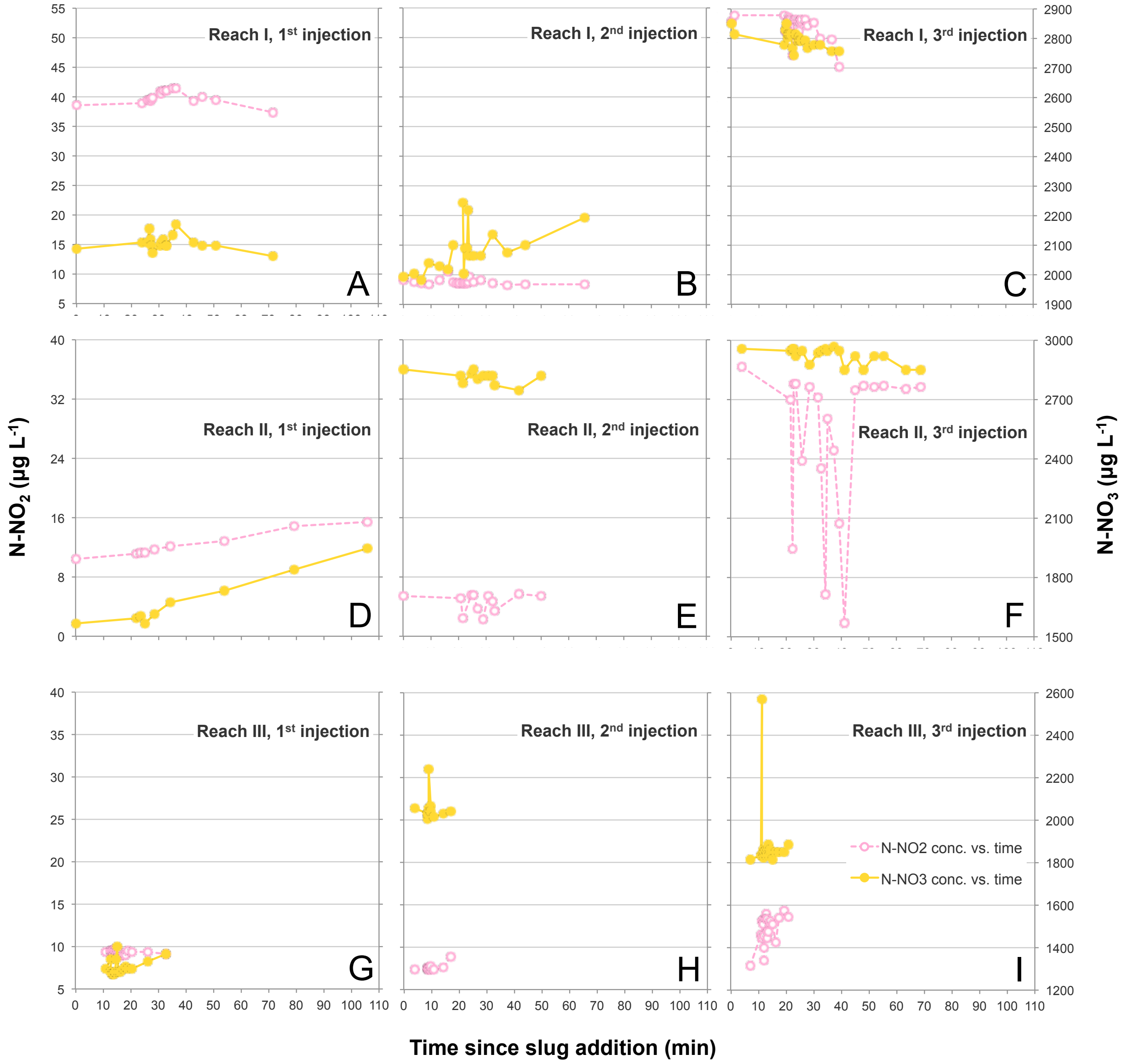


Figure 12 Nitrite and Nitrate concentration (N-NO_2 and N-NO_3, mg L^{-1}) curves during slug passage at reach I of the first **A**, second **B** and third **C** solute injection, at reach II of the first **D**, second **E** and third **F** solute injection and at reach III of the first **G**, second **H** and third **I** solute injection. N-NO_2 data is shown as empty dots and dashed line, N-NO_3 data is shown as full dots and full line.

Table 6 Nutrient spiraling parameter phosphate P-PO_4 and ammonium N-NH_4 results of reach I – III. at the first, second and third injection (inj.). Shown are the estimates of the advection corrected uptake rate at experimental UR_{exp} and at ambient level UR_{amb} (µg m^{-2} s^{-1}), the uptake length S_w (m) and the mass-transfer coefficient V_f (µm s^{-1}). In addition the measured discharge Q (L s^{-1}), the duration of the solute addition experiment (SAE, min), the increase of the chloride Cl background concentration increase (x) and the percentage of Cl retained are listed. Spiralling metrics results applicable for interpretation are high-lightened bold.

Parameter	Unit	Reach I revitalized riffle-pool			Reach II reactivated wetland			Reach III channelized run		
		1st inj.	2nd inj.	3rd inj.	1st inj.	2nd inj.	3rd inj.	1st inj.	2nd inj.	3rd inj.
Q measured	(m^3 s^{-1})	0.16	0.41	0.41	0.24	0.42	0.37	0.41	0.66	0.57
Duration of SAE	(min)	60	38	39	106	47	66	27	16	19
Cl increase	(x)	1.6	1.9	1.3	1.1	1.3	1.2	2.6	2.5	1.9
Cl retained	(%)	35	8	14	81	26	20	9	-20	5
P-PO_4 increase	(x)	1.8	3.9	1.8	1.1	3.1	1.4	24.8	5.8	21.7
P-PO_4 retained	(%)	**37**	**45**	**13**	*30*	**3**	**8**	**10**	**7**	**10**
P-PO_4 U_{exp}	(µg m^{-2} s^{-1})	**0.643**	**0.469**	**0.077**	*0.255*	**0.073**	**0.061**	**0.093**	**0.021**	**0.030**
P-PO_4 U_{amb}	(µg m^{-2} s^{-1})	**0.349**	**0.120**	**0.042**	*0.229*	**0.024**	**0.042**	**0.004**	**0.004**	**0.001**
P-PO_4 S_w	(m)	**4706**	**3491**	**23514**	*5431*	**24946**	**23026**	**55494**	**129253**	**39085**
P-PO_4 V_f	(µm s^{-1})	**10.19**	**16.24**	**2.30**	*6.34*	**3.16**	**2.55**	**0.88**	**0.43**	**1.28**
N-NH_4 increase	(x)	2.8	1.0	1.1	1.6	5.6	1.7	6.09	12.2	3.4
N-NH_4 retained	(%)	**33**	*143*	*104*	*81*	**35**	*64*	**44**	*-13*	*140*
N-NH_4 U_{exp}	(µg m^{-2} s^{-1})	**0.509**	*1.310*	*0.553*	*1.595*	**0.911**	*0.417*	**0.349**	*-0.034*	*0.215*
N-NH_4 U_{amb}	(µg m^{-2} s^{-1})	**0.184**	*1.310*	*0.484*	*0.979*	**0.163**	*0.239*	**0.057**	*-0.003*	*0.063*
N-NH_4 S_w	(m)	**4545**	*504*	*13228*	*424*	**1418**	*3074*	**6084**	*-54215*	*693*
N-NH_4 V_f	(µm s^{-1})	**10.55**	*112.42*	*4.09*	*81.16*	**55.54**	*19.12*	**7.99**	*-1.02*	*72.27*

Mass balance

During the time period of April to October 2002 at low and mean discharges reach I was a sink for $P\text{-}PO_4$, $N\text{-}NH_4$, $N\text{-}NO_3$, DON and DOC (Tab. 7). P_{part} and N_{part} in general were exported and DOP was in steady state. Reach II clearly acted as a sink for $N\text{-}NH_4$ and $N\text{-}NO_3$ as well as for the particulate fractions. The reactivated wetland slightly exported $P\text{-}PO_4$ and further was a source of the dissolved organic compounds DON and DOC. Furthermore results indicated that reach III had been a sink for $P\text{-}PO_4$, P_{part}, $N\text{-}NH_4$ such as for DON and a source of $N\text{-}NO_3$, N_{part} and DOC. The in- and outputs of DOP and were more or less balanced.

The brooks Rotwassergrabenbach and Grünauerbach were not considered in mass balancing as they only contributed in average 1 – 5 % to discharge and the solute and particle loads did not differ mentionable from those of River Wien.

Table 7 Mass balances (MB) results of the parameters phosphate ($P\text{-}PO_4$), dissolved organic phosphorus (DOP), particulate phosphorus (P_{part}), ammonium ($N\text{-}NH_4$), nitrate ($N\text{-}NO_3$), dissolved organic nitrogen (DON), particulate nitrogen (N_{part}) and dissolved organic carbon (DOC) of reach I – III during mean water level from April to October 2002 of River Wien. Shown are the mean, minimum (min), maximum (max) and numbers of dates during the respective reach performed as a sink (positive numbers), a source (negative numbers) or was at steady state.

Parameter	Unit	Reach I revitalized riffle-pool						Reach II reactivated wetland						Reach III channelized run					
		Mean	Min	Max	Sink	Steady	Source	Mean	Min	Max	Sink	Steady	Source	Mean	Min	Max	Sink	Steady	Source
MB $P\text{-}PO_4$	($\mu g\ m^{-2}\ s^{-1}$)	0.23	-0.32	1.03	4	2	1	-0.06	-0.66	0.79	2	0	4	0.12	-0.06	0.44	3	2	1
MB DOP	($\mu g\ m^{-2}\ s^{-1}$)	0.01	-0.15	0.16	2	0	2	0.02	-0.26	0.33	1	2	1	0.01	-0.13	0.14	1	2	1
MB P_{part}	($\mu g\ m^{-2}\ s^{-1}$)	-0.33	-2.37	1.05	2	0	2	0.59	-4.72	7.41	1	0	2	0.06	-0.57	0.73	2	0	2
MB $N\text{-}NH_4$	($\mu g\ m^{-2}\ s^{-1}$)	6.24	-0.61	27.46	6	0	1	0.99	-0.51	4.86	3	1	2	0.39	-0.13	1.53	5	0	2
MB $N\text{-}NO_3$	($\mu g\ m^{-2}\ s^{-1}$)	7.58	-9.41	59.94	5	0	2	6.58	-32.69	91.12	3	0	3	-1.18	-8.54	6.45	3	0	4
MB DON	($\mu g\ m^{-2}\ s^{-1}$)	1.90	-1.81	9.41	2	0	2	-1.12	-7.07	1.83	2	1	1	0.36	-1.53	2.56	2	1	1
MB N_{part}	($\mu g\ m^{-2}\ s^{-1}$)	-2.78	-11.61	3.13	2	0	2	0.76	-3.93	4.85	2	0	2	-1.72	-8.07	2.41	1	0	3
MB DOC	($\mu g\ m^{-2}\ s^{-1}$)	9.96	-20.40	33.92	3	0	1	-34.75	-84.62	9.45	2	0	2	-16.94	-51.53	3.28	1	0	2

DISCUSSION

Riverbed structure

The riverbed structure shows vast effects on water velocity, strongly controls in-channel transient storage and is of importance for nutrient spiraling (Gooseff *et al.* 2005; Bukaveckas 2007; Bottacin-Busolin *et al.* 2009; Hoellin *et al.* 2009). Channel morphology and sediment composition influence hydrological connectivity, biofilm structure, epilithic algal growth, macrophyte emergence and organic matter deposition (Harvey *et al.* 2003; Bottacin-Busolin *et al.* 2009; Hoellin *et al.* 2009). Channel complexity enhances water residence times and supports biogeochemical solute processing (Doyle *et al.* 2003; Gücker & Boëchat 2004; Gooseff *et al.* 2007). In addition, riparian vegetation adds to the maintenance of steady water temperatures, nutrient uptake from surface as shallow ground waters as well as towards important habitat-forming processes through delivery of woody debris (Tabacchi *et al*. 2000; Roberts *et al*. 2007).

Channel design is the basal component of river restoration (Bukaveckas 2007). Riverbed re-structuring was a prime objective for restoration engineering at the urban River Wien (Goldschmied 2002). Frame conditions at River Wien allowed a moderate revitalisation. Bedrock, housing and traffic development limited the possibilities of restoration. Flood-control had to be ensured; sediment movement and the reclaimable area were constricted. As no data previous of the restoration was available, data of the channelized run were treated as of 'pre-restoration' and of the reactivated wetland and the revitalized riffle-pool as of 'post-restoration' state (according to Bukaveckas 2007).

Effects of different restoration measures on channel geometry and surface sediment composition showed a distinct ranking of complexity amongst the restructured reaches and in comparison to the channelized reach. Integration of River Wien in the flood-control reservoirs (FCRs) enabled a widening of the riverbed, the establishment of a riffle-pool sequence and the accumulation of debris dams – all supportive to channel complexity. The riverbed of the revitalized riffle-pool section was constricted, limited in broadening (both sides bordered by an inclination and at the left bank bounded by a path, same as at the channelized section). Although the restructured reaches were similar in mean width the variation of width at the reactivated wetland was much larger, reflected in greater sinuosity. CV width had strongly contributed to the high channel complexity at reach II. Most likely the high proportion of smaller grain sizes supported erosion and formation of side-pools. Output of large cobbles in the FCRs had been negligible, for riverbed stabilization was non-essential.

In consideration of the fact that the upper third of reach II exhibited widths of up to more than 17 meters, we presume sinuosity of this section to have been underestimated. The outsourced d-GPS survey, for obtainment of the shoreline length, most likely had not taken into account the wetted area of reed of the first transects, flooded even at low water levels. Nonetheless sinuosities were in agreement with published data: Sinuosity indices for the step-pool (1.3 ± 0.2) and run (1.1 ± 0.2) morph-type in the Serra do Cipó National Park (Brazil; Gücker & Boëchat 2004) were in range with values of the restructured and channelized reaches at River Wien, as well as sinuosities of the agricultural (1.26, Headquarters Stream), the reference (1.39, Ditch Creek) and the urban section (1.02, Jackson Hole Golf Stream) at the Experimental Reaches in Jackson Hole (Wyoming, USA; Gooseff *et al.* 2007).

Sediment at reach I was well mixed, due to the output of gravel and cobbles as riverbed stabilization-elements. Sediments of reach II, in comparison, were more sorted despite the excavation of over the intervening years deposited fine sediments. Sediment movement and mixing in the FCRs seems to be restricted to high floods as well as replenishment of material and affected by the position of weir-shutters. Rich organic matter accumulations at the reactivated wetland derived from the dense surrounding reed belt. Deposition was supported by reduced flow velocities due to the occurrence of pools and eddies. In 2005 Uhlik estimated sediment grain sizes at the site of reach II. Q75 was 33 ± 12 mm, half the size of estimated in 2002 by the present author, clearly indicating occurring sediment dynamics (personal communication, published in Fuchsberger 2009, p. 10). Most likely flood events shifted sediments and deposited smaller grains.

Result of riverbed unification at the channelized run was extensive geomorphological degradation. Reach III the most anthropogenic-influenced section was the least complex. Grimm *et al.* (2005) found channel complexity depleted in urban streams. Alluvial deposited sediments rested on the flagstones in the upper tenth of the reach. Strongly exposed to friction they hardly offered a habitat for algae. In turn, downstream immobile flagstones proved to be great growth-grounds for filamentous algae. More than 40 % of the stream-bottom was covered, resulting in highest chl-*a* concentrations per square meter amongst the study reaches.

Formation of eddies, riffles and pools as well as mixing of sediments pointed out how beneficial in terms of riverine ecology restoration of River Wien and its integration in the FCRs was. Besides a general rise in channel complexity, bioindicators gave further evidence on a gain in habitat heterogeneity. Restored sections held habitat improvements for macrozoobenthos (Katzmann & Forster 2002, p. 34) and distinct increases in fish densities and species numbers (Keckeis 2002, p. 49). Facilitated flow continuum and hence matter dynamics on behalf of opened FCR weir-shutters up to a discharge of 30 $m^3 s^{-1}$ supported reestablishment of structure-elements and widening of riparian buffer zones necessary for settlement of rheophil dragonflys (Raab 2002, p. 62).

Solute addition experiment

Total transient storage

The importance of total transient storage in fluvial ecosystems concerns the storage and motion slowdown of the water. Rough porous streambeds fostering interstitial flow, high ratios of streambed area to channel volume as well as tight riffle-pool sequences and occurrence of eddies decelerating surface water-flows have been observed to promote in-channel and dead-zone storage (D'Angelo *et al*. 1993; Alexander *et al*. 2000; Gooseff *et al*. 2007). Not missing mentioning slope, interplay with adjacent compartments, flow obstructions and stream communities crucial for hydrological retention (Newbold *et al.* 1982; Harvey & Wagner 2000; Ensign & Doyle 2005; Stofleth *et al*. 2008).

Transient storage is clearly impacted through streambed restoration, as dependency upon channel-morphology and bed-sediments is strong (Gooseff *et al*. 2007). Reciprocal assessment of transient storage is helpful to examine effects of restoration measures on riverine ecology. Achieved benefits of riverbed restoration are the decrease in flow velocity and increment in transient storage (Bukaveckas 2007). At the River Wien, in terms of the extreme hydrophobic property of the bedrock in the catchment area, occurrence of transient storage improves local hydrology (Palmer 2008) and may be of relevance for flood protection, underlining the importance of the restoration.

Slug injections as the one-dimensional model applied for the estimation of transient storage were feasible to tackle questions of restoration effects at River Wien. Calculation on the basis of the conductivity run was simple and firm, thanks to distinct conservative tracer peaks achieved in most cases. On the contrary, calculation of nutrient spiraling on the basis of nutrient concentrations was more complex and instable.

Relating to the physical template, results indicated that improvements in geomorphic form were of primary importance for transient storage. Correlations revealed variation of river width and depth very likely being the predominant factors regulating transient storage. These observations were in agreement with Gooseff *et al.* (2007), which found channel width variation most influential on in-stream and dead-zone storage. Transient storage metric-ratios of the three study reaches in general showed high congruence with computed values of channel complexity. Gooseff *et al.* (2007) as well identified geomorphic complexity in general to extend water residence-times. Median travel time at a restored segment of Wilson Creek (KY, USA) was half of at a channelized section (Bukaveckas 2007). As variation of width was most important on channel complexity major transient storage performance was found at the reactivated wetland, where complexity was greatest (Fig. 9). Pronounced existence of riffles and pools at reach II, reflected in the highest value of CV depth amongst the study sections, supplementary supported transient storage. The fact that the magnitude of retention compartments at the restructured reach I was rather similar to the channelized reach indicated that restored structural conditions at River Wien did not promote areal extension of *TSZs* (transient storage-zones) to the extent as hydrological exchange was facilitated. Regardless percentages of storage area, results of A_S/A (water exchange rate from the water-column to the *TSZ* and vice versa) showed amplified occurrence of transient storage at reach I compared to reach III. On grounds of diminished geomorphic complexity reach III exhibited smallest transient storage. Both hydrologic exchange rates K_1 and K_2 were elevated. T_W (water travel time in the water-column before moving in the *TSZ*) and T_S (water molecule average time in the *TSZ* before backflow to the water-column) reflected water entering and leaving the *TSZs* rapid. Flagstone sealing presumably hardly allowed for any hydrologic connectivity. Besides the porous volume of sparsely deposited pebbles, only small-area break-offs of concrete jointing (or possible cavities) could have offered additional areas for hydrological retention. Concerning the restructured sections, surface-sediment composition presumably was of secondary, and heterogeneity was of least, importance for transient storage. Well-mixed sediment at reach I probably did not facilitate water-exchange from the free-flowing water to the *TSZ* and water left the storage zone proportionally faster, resulting in smaller ratios of A_S/A and T_W/T_S in comparison to reach II. At the reactivated wetland high percentage of silt and sand, inducing a greater effective pore-volume (Hölting & Coldewey 2009) of reach sediments, might have enhanced flow from the water column to, and prolonged stay of water in, the storage zone.

Concerning the biogenic realm the present author assumes richness in organic matter to have contributed to transient storage, overall at the reactivated wetland section. Biofilms, plants, wood- and leaf-litter flow obstructions and macro-zoobenthos very likely were of relevance for increase in surface roughness. According to Stofleth *et al*. (2008) flow obstruction caused by large wood and debris dams induced total hydraulic retention by 50 – 100 % at a small sand-bed stream. Mulholland & Webster (2010) advection-dispersal model exhibited periphytic biomass to expand *TSZs*. Driven through geomorphology biofilm mass was not only found responding and biofilm architecture adapting to the hydrodynamic environment but substantially adding to transient storage (Battin 2000; Battin *et al*. 2003; Bottacin-Busolin *et al.* 2009). River Wien streambed restoration very likely supported growth of biofilm *TSZs*. At the channelized section dense mats of filamentous algae functioned as living *TSZs*. Possibly in parts algae facilitated in-stream transient storage, although high velocities indicated that on the whole they could not compensate the lack of flow-resistance elements.

In the present study flow velocity had lesser effect on transient storage than channel complexity. D'Angelo *et al*. (1993) found A_S/A as well to increase with in-stream channel complexity, but on the contrary decrease with rise in velocity. Flow velocity besides determined by discharge expressed strongest dependency on sinuosity.

Increase in sinuosity contributed to flow retardation as well in the studies of Gooseff *et al.* and Bukaveckas (2007) respectively. Greater proportion of large cobble at reach I (Q75 was the double of at reach II) seemingly did not support velocity slowdown, in terms of serving as flow obstruction.

River Wien A_S/A results were within the size range observed at rivers of 4^{th} to 5^{th} order. Estimates of A_S/A ranged from 0.53 to 0.09, at unconstrained and constrained 5^{th}-order sites in the US Cascade Mountains (D'Angelo *et al*. 1993). Paul & Hall Jr. (2002) received similar values for A_S/A (0.18 and 0.19) at 4^{th}-order Upper and 5^{th}-order Lower Hubbard Brook. Discharges at Hubbard Brook were three to six times smaller than at River Wien, additionally supporting the finding of transient storage being largely insensitive to discharge. Investigating geomorphologic diverse Brazilian tropical headwaters Gücker & Boëchat (2004) detected a wide range for A_S/A. Significant differences between stream morph-types (step-pool, run and swamp) in all hydraulic parameters demonstrated channel morphology control on transient storage.

Literature reports have exhibited both the flow-channel and the hyporheic compartment contributing to total transient storage. Ensign & Doyle (2005) found in-channel transient storage to be of greater importance for hydrological retention than hyporheic storage. Mulholland *et al*. (1997) and Hall *et al*. (2002) reported on hyporheic storage to be the significant component for matter retention. In the study of Stofleth *et al.* (2008) only 0,01–0,49% of total exchange accounted for hyporheic storage, and did not rise subsequent to installation of flow obstructions. Ordinary tracer experiments do not differ between compartments contributing to transient storage (Harvey & Wagner 2000), nor give evidence to what extent vertical and lateral hydrological connectivity occurs. At River Wien restructured reach I probably surface-water transient storage outweighed hyporheic storage. Channel geomorphology and marginal sediment permeability (Fuchsberger 2009, p. 30) supposedly impeded expansion of the hyporheal. At the reactivated wetland presumably hyporheic exchange supported transient storage to small parts. The study of Fuchsberger (2009, p. 30) in 2005 on hyporheic exchange in FCR basin 6 indicated that 39–97 % of the hyporhean water was of surface-water origin. Stream hydrograph and ground water levels mainly controlled hyporheic exchange (Fuchsberger 2009, p. 31).

Estimation of total transient storage proved to be utile for appraisal of restoration success and quantification of in-stream hydrologic processes. Increase in channel complexity at both restructured sections favored a clear ascent in transient storage. Flow velocity and downstream movement of water were decelerated, confirming positive effects of riverbed restoration. In addition restoration measures at River Wien very likely promoted lateral and vertical flow dispersal supplying the riparian belt with water and rejuvenating the groundwater body as well as created adjacent terrestrial retention areas.

Nutrient spiralling

Spiraling of the metabolically important elements phosphorus (P) and nitrogen (N) are cornerstones of ecosystem biogeochemistry (Allan 1995; Ensign & Doyle 2006). Research on nutrient spiraling has proven to deepen the comprehension of in-stream processes as of the transformation of P and N inputs (Meyer & Likens 1979; Mulholland & Webster 2010). Applied conservative tracer and nutrient release experiments identified geomorphological channel features, grain size, flow velocity, transient storage, background concentration, sediment chemistry and biological activity as controlling elements of nutrient retention (Munn & Meyer 1990; Martí & Sabater 1996; Valett *et al*. 1996; Alexander *et al*. 2000; Dodds *et al*. 2002, Hall & Tank 2003).

Stream restoration has been found to stimulate nutrient dynamics and mitigate downstream matter transport (Bukaveckas 2007; Klocker *et al*. 2009; Weigelhofer *et al*. 2013). A more complex streambed encourages the prolongation of water residence time, the rise in benthic area to channel volume and interstitial flow thus providing opportunities for accelerated nutrient uptake and transformation (Newbold *et al.* 1982; Allan 1995; Fisher *et al.* 1998; Gooseff *et al*. 2007; Palmer 2008). Solute addition experiments (SAEs) revealed differences in spiraling properties of restructured versus heavily modified streams informing on ecological processes occurring, despite all methodological limitations (Bunn *et al.* 1999; Ensign & Doyle 2006; Bukaveckas 2007). Moreover, SAEs operating at the reach spatial scale, contrariwise to water quality point sampling, may be the preferred approach towards valuation of ecosystem health (Boulton 1999).

At River Wien strong daily fluctuations in hydrology and nutrient concentrations complicated the calculation procedure and interpretation of spiraling results. Allan (1995) commented local human activity, rainfall, season and position along the river continuum to cause variation in solute background concentration. Concerning the present study strong seasonal variability as well as occurrence of oxidation processes was the case. Multiple failures to attain proper increases came about. Overall ammonium was experienced frail to the circumstances as the methods applied (and is further left almost uncommented, by reason of poor data). Besides, in cases of greater hydrological than biogeochemical nutrient retention (evident at two additions) no spiraling data could be gained. Temporary limited availability of laboratory facilities and the aim to stay within the requirement profile of a masterwork resulted in not realizing supplemental injections. All mentioned handicaps lead to a decimated outcome. Therefore detailed questions addressed could not be answered sharp. Nevertheless a snapshot of River Wien restoration effects on nutrient spiraling can be offered and conservative statement given. Further, as the metrics uptake rate at experimental level (U_{exp}) and spiraling length (S_W) are sensitive to discharge, a direct result comparison only was permissible amongst the restructured reaches similar in discharge. For comparison with the channelized reach as with spiralling findings of selected literature V_f was considered. V_f has advantage over U_{exp} and S_W

as it corrects for various depths, flow-velocities and concentrations amongst streams and is frequently documented (Ensign & Doyle 2006; Newcomer-Johnson *et al.* 2016).

More than two decades of SAEs have shown that stream nutrient retentiveness to a great extent is a matter of physics-based hydrodynamics (Stream Solute Workshop 1990; Newbold *et al.* 1992; Vallet *et al.* 1996; Ensign & Doyle 2005; Lautz & Siegel 2007; Argerich *et al.* 2011). Transient storage is of primary importance for nutrient spiraling, since the impact of biota on total transport is proportionally small (Newbold 1992). Given that transient storage offers longer contact-times with reactive surfaces, it both regulates availability and activates biogeochemical processing of nutrients (Butturini & Sabater 1998; Decamps 1996; Vallet *et al.* 1996 and 1997; Pusch *et al.* 1998). Several studies reported on S_W shortened and rates of nutrient uptake and mass transfer (V_f) raised in streams holding extended *TSZs*, rich in quiescent water, organic matter and biota: In the study of Gücker & Boëchat (2004) half of the ammonium uptake took place in surface *TSZs*, at all reach morph-types of Rio Cipó watershed. Butturini & Sabater (1999) observed a strong tie between *TSZ*-size and phosphate and ammonium retention at a 2nd-order sandy-bottom Mediterranean stream. Research of Mulholland *et al.* (1990) identified shorter P transport lengths associated with transient storage. 35 % of ammonium variation during summer months was explained by transient storage in the survey of Hall *et al.* (2002); phosphate uptake unlike was not transient storage related, presumably because P was to large parts chemically adsorbed. Results of Newcomer-Johnson *et al.* (2016) showed that soluble reactive phosphorus (SRP) uptake was significantly related to the watershed area and SRP concentration; ammonium spiraling metrics correlated with flow velocity and transient storage. Velocity slowdown, as an effect of channel restoration at Wilson Creek, augmented the first-order uptake rate coefficients of P and N by the 3- and 30-fold within the restructured section in comparison to its channelized condition (Bukaveckas 2007). Simon *et al.* (2005) found changes in stream depths and velocities accounting for variations in P and N S_W, however transient storage (as well as temperature and chl-*a*) was a weak driver of uptake. Furthermore spiraling results of P and N fairly correlated, pointing at various influencing parameters accountable for either uptake and that natural within-stream variance of nutrient uptake can be large (Simon *et al*. 2005). Literature on the tie between transient and nutrient storage remains partially controversial (Argerich 2011). Nevertheless the global review of Newcomer-Johnson *et al.* (2016) on nutrient retention in restored streams and rivers identified a clear relation amongst watershed attributes (such as area, solutes concentrations and transient storage) and uptake.

At River Wien base-flow conditions combined effects of restored channel morphology, decelerated flow velocities and enhanced transient storage likely had positive impacts on the spiraling of phosphate, although no correlates with transient storage metrics were identified. Possibly chemical adsorption determined P retention at the restructured reaches I and II. According to Newbold *et al.* (1982), who found the downstream velocity of the particulate compartment to influence spiralling, the present author argues that the decreased water velocity at restored River Wien encouraged deposition of suspended solids, particularly at the wetland section, and hence supported nutrient retention. At the channelized reach in contrast transient water and solute storage zones were of relevance for the efficiency of P uptake. A correlate was found between S_W and A_S/A (R^2 0.999) as well as with the size of the *TSZ* (R^2 0.976).

Surface-subsurface exchange strongly favors nutrient interaction with sediments and the biocoenosis, giving greater chance to retention and for recycling (Allan 1995; Dahm *et al.* 1998; Palmer 2008). Overall near-surface sediments have been of importance for abiotic retention mechanisms and utilization of phosphate and ammonium (Allan 1995). Butturini & Sabater (1999) found the upper ten sediment centimeters adding more to phosphate detention than those at 25 cm depth; further, ammonium was completely absent at any hyporheic depth, indicating sole rapid benthic take up. At the River Wien reactivated wetland clogged sediments restricted downwelling to few areas (Fuchsberger 2009, p. 33). The core of in the sediment-injected solutes (phosphate more than ammonium) were retained at the surface, implying that also in FCR 6 the hyporheic was of lesser importance for nutrient cycling (Fuchsberger 2009, p. 21 and 33). Perhaps reduced flow velocities favored seston deposition and in consequence colmatation of effective-sediments. The mentioned study outcome suggests in-channel above hyporheic transient storage to have supported nutrient retention in the presented SAE.

Physical-geochemical processes have overall shown effect on phosphate, and to a minor degree on ammonium, concentrations (Allan 1995). Precipitation and sorption of phosphate ions to very fine inorganic and colloid organic particles (Allan 1995) may have in parts accounted for P removal at River Wien restored sections. Overall at the reactivated wetland sediment composition, such as high portions of small sized grains and organic matter stocks, provided enlarged reactive surfaces contributing to adsorptive nutrient retention. Sediment chemistry analysis indicated sediment-bound phosphate-immobilization to have played a relevant role. Weighted sediment total phosphorus concentration at reach II (Hein 2002b, p. 13) was the triple of at the restructured reach I. As aerobe sediment conditions facilitate complex forming of phosphate with iron or aluminium oxides and magnesium hydroxide coatings of charged clays redox-potentials appropriate for P binding (Allan 1995; Gabriel *et al.* 2006) likely occurred at River Wien due to the condition of the water column well saturated with oxygen. At Hubbard Brook Experimental Forest geochemical sorption reactions even were of higher significance for the immobilization of dissolved inorganic P (DIP) than biotic uptake and transformation (Meyer & Likens 1979). Adsorption to sediments of DIP, and during summer months of $N\text{-}NH_4$ as well, substantially was enhanced after experimental enrichment at Bear Brook (NH, USA, Richey *et al.* 1985). And at Ledbetter Creek DIP adsorption, as well as microbial de- and nitrification, took mainly place in the fine-grained sediments of deeper pools (Jin *et al.* 2007).

In streams biotic withdrawal of nutrients from the water column is primarily a benthic process (Sabater *et al*. 2002). Autotrophs, heterotroph microbes and their consumers to large parts realize solute intake proximal and within the streambed sediments (Comte & del Giorgio 2010). Organic rich sediments are a potential source of C, P and N (Allan 1995). Therefore abundant benthic detritus is a hotspot for biofilm growth by means of strong effect on bacterial metabolism (Groffman *et al*. 2005; Palmer 2008; Hoellein *et al*. 2009). Particularly biofilms are prime sites of nutrient digestion directly inducing rapid uptake (Valett *et al*. 2002; Battin *et al*. 2003; Bottacin-Busolin et al. 2009). Literature indicates that microbial processes as temporary storage in biotic compartments (e.g. as reed, algae and zoobenthos) most likely controlled ammonium spiraling sizable at River Wien. Observed increases in N-NO_2 and N-NO_3 during the SAE gave evidence toward likely nitrification of parts of the ammonium added. Newbold *et al*. (1983b) in their continuous ammonium enrichment study detected nitrification and in the addition experiment of Ashkenas *et al*. (2004) nearly half of the $^{15}NH_4Cl$ released was exported mainly in form of $^{15}NO_3$. A general good nutrient supply as well as possible occurrence of nitrification suggest nutrient demand was low and may explain small uptake velocities during the present SAE. V_f of SRP and N-NH_4 were within the lower range of published study results (e.g. Simon et al. 2005; Bukaveckas 2007). As production is resource driven excess P and N relative to demand may enhance productivity rates but assimilatory uptake proceeds slower and transport lengths are stretched (Ensign & Doyle 2006; Allan & Castillo 2007; Leitner 2016, p. IV). In addition, P-PO_4 V_f at the restored reaches interestingly was in average the eight-fold of at the channelized reach suggesting that the restoration enhanced biogeochemical P retention.

In urbanized streams nutrient removal from the water column has been observed small and slow (Grimm *et al*. 2005; Meyer *et al*. 2005). Decline in uptake rates as in uptake velocities of both phosphate and ammonium were directly related to increasing catchment coverage, altered hydrology as well as losses in channel complexity and benthic organic matter (Meyer *et al*. 2005). Estimated theoretical average uptake lengths indicated that at the River Wien heavily modified section a nutrient molecule might not even once have the chance to accomplish a spiral – being taken up and transformed once during it's journey downstream. Presumably complexity loss above all had implications on nutrient spiraling. However filamentous algae extensively covering the channel bed, stripping nutrients from the water column, assumedly were supportive to spiraling. Biofilm coating of algae belike added to nutrient consumption. Sand-Jensen *et al*. (1989) discussed high densities of filamentous algae to may have boosted ammonium uptake at a channelized reach. In the study of Martí & Sabater (1996) conducted at Riera Major algal activity primarily controlled phosphate retention. Riparian deforestation at Riera Major increased biomass growth as well as algal surface coverage and subsequently enhanced the retention efficiency of phosphate by far than the twofold (Sabater *et al*. 2000). Stimulatory effects of direct radiation, on behalf of absent riparian shading, might as well have contributed to algal production adding to nutrient uptake at the River Wien channelized section.

In this context it is important to pin point the potential significance of flash floods on standing stock of periphyton, detritus and microorganisms. Community status is therefore dependent on the timing since the last flooding (Fisher *et al*. 1982 and 1998; Grimm 1987). This may have played a role for observed uptake patterns found during the present study after the mid-August flood throughput. Referring to literature presumably to most parts biomass had recovered and growth had levelled off prior to the second solute injection.

Closing the discussion on the aspects of nutrient spiralling the present author attempts to address if the restoration measures had amplified River Wien capacities for nutrient uptake – as defined by Minshall *et al*. (1983) wetlands react conservatively to nutrient additions, thus providing trophic stability to fluvial systems. Surprisingly observed differences in P-PO_4 (and N-NH_4) U_{exp} and U_{amb} indicated that all three study reaches had held free abiotic retention capacities or biotic nutrient demand and that the SAE had stimulated uptake. Interestingly ratios of U_{exp}:U_{amb} were identical with increases in background values. This observation suggests mass transfer as enzyme kinetics at the respective SAE dates to have been within linear progression of the Michaelis Menten relation. Even at background increases exceeding the 20-fold saturation apparently had not occurred. This notice is in contrary with several recommendations not to overshoot background by the double, due to risk of system saturation (Thomas *et al.* 2003; Newbold *et al.* 2006; Arango & Tank 2008; O'Brien *et al.* 2008). Nonetheless Dodds *et al*. (2002) neither found uptake limited during high increments – uptake rates responded to rises in N-NH_4 concentrations indicating that U_{exp} may not necessarily saturate in streams experiencing short-term peaking of downstream nutrient fluxes (e.g. as reaction to flood events). In a ^{15}N addition study O'Brien *et al.* (2007) did not find uptake to saturate with increasing nitrate concentration – a finding confirmed in the LINX II study (Mulholland *et al*. 2008). Furthermore it is compulsory to consider the possibility that estimates of U_{exp}:U_{amb} were consequence of the model applied, a function of background increase (background and peak values were part of the U_{amb} calculation). In that case U_{exp}:U_{amb} was not an indicator for biogeochemical P and N uptake potentials nor system saturation. Hence data did not support answering if the reactivated wetland section would have exhibited the highest range of uptake capacity during elevated nutrient conditions.

Summing up, dimensions of transient storage and nutrient spiraling mirror the states of physical and biogeochemical stream-properties. Not only did restored streambed conditions augment transient storage and the likelihood for nutrient spiraling, furthermore the amplified River Wien ecosystem functioning is a regulative to self-purification. As wetlands represent hot sites of matter processing (Mitsch & Gosselink 2015) very likely the reactivation of the FCRs contributes extraordinarily to the retrieval of P and N. All in all conducive to water quality as river health, easing implications of urban land use practices (e.g. urban stream syndrome; Pickett *et al*. 2001; Palmer 2008).

Mass balance

Mass balancing is the common method to estimate the function of a river system (or section) as a source or sink of matter. Within the framework of the project "Hydrochemistry, suspended solids dynamics and organic pools of the River Wien" mass balancing was used on a regular basis to monitor effects of restoration efforts on matter dynamics. The mass-balance approach is easy in application and demands little time in the field as with regard to the calculation mode, in comparison to a SAE. The SAE on River Wien was complementarily introduced to the monitoring project as a promising though laborious method challenging to grasp detailed aspects of matter retention.

Short-term mass balances of the vegetation period 2002, as reported, revealed SAE reach I to have been a substantial sink for $P\text{-}PO_4$ and $N\text{-}NH_4$ (interestingly reach III as well, although to minor extent). At the reach scale leaf litter such as plant leaching or consumer excretion may have been greater than $P\text{-}PO_4$ retention at the reactivated wetland. Nevertheless taking into account all five River Wien FCRs 20 % of $P\text{-}PO_4$ riverine input was retained in 2002 (Hein 2002b, p. 3). Rough estimates of mass-balance nutrient fluxes per square meter more or less coped with U_{amb} results from the solute-additions, thus indicating that methods of mass balancing and nutrient spiralling were overall congruent. Furthermore the mass-balance approach gave indications for sink, source or steady state solute system patterns. Nutrient spiralling always displayed retention, even if retention was small. Consequential nutrient uptake occurred even if in total matter export took place.

Long-term mass-balance residuals allowed for more distinct statements on the retentive character of the River Wien FCRs: by way of example, on the mesoscale from weir 1 to 5 (2001 – 2005) inputs of $P\text{-}PO_4$ (10 %), P_{tot} (14 %), $N\text{-}NH_4$ (39 %), N_{tot} (13 %) and TSS (31 %) were effectively retained. The function of the FCRs as a retention body for particles (Weigelhofer *et al.* 2005, p. 19 and 20) reflected effective transient storage in this man-made wetland.

As nutrients principally are associated with fine particles (Allan 1995) occurrence of clay lenses, silt and sand deposits rich in POM at the FCRs promoted adsorption of phosphate to mentioned particle surfaces. 75 % of the phosphate pool retained in the FCRs was found adsorbed to the sediment (W1 – W4, 1999 – 2001; Hein 2001, p. 32). During flooding of the extended areas of reed, plant structures very likely supported particle sedimentation and the FCRs role as a water purifying bioreactor. However plants temporarily seem not to store the bulk of nutrients streaming past, since rather algae and bacteria resemble the major sinks for inorganic nutrients in most fluvial systems (Allan 1995). Invertebrate consumers migrating from the habitat add to nutrient removal from the system (Kitchell *et al.* 1979). With reference to literature (Cooper & Cooke 1984; Martin *et al*. 2001; Seitzinger *et al*. 2006, Alexander *et al*. 2009) benthic microbial mediated processes, fostered through high N-NO_3 levels of River Wien as presence of organic debris, seem to have relevantly contributed to heterotrophic respiration at the FCRs thus potentially enhancing denitrification-rates and N_2 out-gasing respectively. Denitrification depleted N-NO_3 charges by 15 % during summer baseflow at River Dorn, England (Cooke & White 1987). In a study of Roberts & Mulholland (2007) mass balancing indicated that substantial in-stream DIN retention corresponded to spring primary production as well as to heterotrophic respiration linked with autumnal leaf decomposition. An experimental approach using in-vitro flow bioreactors demonstrated that during short-time N-NH_4 additions nitrification rates of the River Wien FCR microbial community increased by the double and the potential N-NH_4 uptake capacity approximated 87 % of peak inputs (Glaser 2001, p. 13; Hein 2001, p. 40).

The reintegration of River Wien in the entire FCRs activated critical stream ecotones such as riparian floodplain and hyporheic zones, confirmed in the long-term trend, certainly being of extraordinary importance for matter retention (Hein 2001, p. 32 and 33, Hein 2002a, p. 18). As supplement for simple mass balances the SAE study on River Wien provided limited additional information on matter dynamics on behalf of mentioned result losses. Nonetheless V_f and S_W gave a first rough insight on how fast and how efficient uptake took place, in addition to U_{amb} and mass balance results on how many nutrients were processed in a respective habitat. SAEs in general do offer ancillary information for management concepts of riverine rehabilitation-projects. S_W e.g., as a function of channel complexity and discharge, may serve comprehension on the required section-length to be restored in order to achieve close to optimum nutrient retention.

Technical experiences and implications for practice

Suggestions for further research on assessment of the riverbed structure and the set-up of solute addition experiments

The high significance of the riverbed structure in the analysis on transient storage and nutrient turnover implies a more detailed assessment of channel morphology and sediment composition in future studies: Experience from the present SAE showed that, in addition to a general raise in river width and depth recordings and sediment sample numbers, detailed measurement of the riffle-pool sequence and evaluation of sediment porous volume would have provided a profounder inspect on the bed structure of the restructured river sections - supportive to more distinct estimates on channel complexity and of interest for conclusions on effects of restoration efforts.

Literature on solute addition studies documents uptake distortion and saturation effects, on behalf of too intense enrichments (Newbold *et al.* 1982, O'Brien *et al.* 2008, Thomas *et al.* 2003). Recommended are slug concentrations increasing tracer backgrounds by the 1.5 and nutrient values by the twofold. Regarding the present SAE in cases of large background increases saturation effects were not observed, as noted previously in the nutrient spiraling discussion section. High precise peaks, as well as clear curve limb in- and decreases, rather supported stability of the model applied and reliability of the data obtained. Challenges experienced, on the contrary, were underestimation and drifting of background values. We observed daily Cl, $P\text{-}PO_4$ and $N\text{-}NH_4$ concentrations vary by far from the mean of the previous years (which was the basis for slug carboy estimates). High standard deviations yet had indicated that this could happen. Daily solute concentrations below or above the expected resulted either in elevation of more than the double or in insufficient background increase. Lack of distinct peaking impeded spiraling parameter estimation and background drifting caused result distortion. Correction for background drift was a necessity. In general for SAEs to be realized in oligo- to mesotrophic systems showing a wide range in solute standard deviations experience from River Wien indicates that a more generous carboy addition aiming at a suggested four-fold background increase could be of service. This recommendation however does not apply to eutrophic systems as they could be close to saturation.

This paragraph briefly sums up basic practical teachings learnt, offered as supportive remarks for all students interested in or planning solute additions: First, choose sites comparable at all levels. Second, never go for a samples number (*n*) of 3. This is of too high risk. With an *n* of 3 you're just on the very edge of makeable statistics and a reliable interpretation is doubtful. For obtainment of hard fits it is essential to increase sampling numbers overall in heterogeneous and complex urban river systems. Also if for example Bukaveckas (2007) stated that "Transient storage was quite uniform over a range of discharge conditions such that only 3 to 4 experiments were required to yield an average value within 10 % of that attained from all experiments (10+) at a given site." At River Wien greater numbers of injections are severely needed. Third, before starting in field and lab make sure to have the appropriate methods to answer you questions, know your method well and thoroughly know how to calculate your results. Don't continue from one step to the next is something has been left unclear. Forth, immediately after obtaining data, check your results. Clarify what has worked out smooth or if improvements are necessary. And fifth, use field testing-kits to get a rough idea of background values. If current solute concentrations are higher than expected add more nutrient salt to the slug carboy in the field.

Regarding possible further research on matter dynamics at River Wien a solute addition study on in the meantime rehabilitated heavily modified segments would be of considerable interest. Direct comparison with remaining channelized sections (alike in discharge) permits capture of pre- and post-restoration states of transient storage and nutrient spiraling.

CONCLUSION

Remediation measures implemented at River Wien appreciably downsized consequences of urbanization on riverine functional attributes. Augment in channel complexity exhibited strong positive impact on hydrologic properties. Particularly establishment of a riffle-pool sequence and variation in river width were of crucial importance for total transient storage. Although restoration seemed not to have resulted in significant gains of storage zone sizes, water residence times clearly increased. Findings of the revitalized riffle-pool and the reactivated wetland in comparison with those from the channelized section revealed both a distinct improvement as well as opportunities for prospective best-management practices. Furthermore solute addition experiments (SAEs) affirm a worthy application in quantifying effects of riverbed restoration on hydrodynamics.

Considerable retention of P as N within the River Wien restructured flow sections pointed at benefits of enhanced physical and biogeochemical conditions. Decelerated flow, enlarged hydrological exchange within transient storage-zones and ample debris structures advanced nutrient processing and hence were regulative to the capacity of riverine self-purification. Reintegration of the main channel within the wetlands of the flood-control reservoirs and their central function as a sink for anthropogenic inputs resembles a major melioration of the ecological status of River Wien, as “phosphorus retention is considered one of the most relevant attributes of natural and constructed wetlands” (Mitsch & Gosselink 2015). Restricted outcome of the SAE on nutrient spiraling implicated no clear evidence towards potentials of matter dynamics at River Wien. Profound comprehension and review of the nutrient cycling model applied are mandatory. Beyond that this research work may provide additional expertise on prospects and challenges of SAEs.

Channelized run sections represent hydraulic-engineering practices out of date. To alleviate adverse repercussion of remaining heavily modified segments on river functioning in the highly urbanized area of Vienna city calls for further remediation of the riverbed as of former riparian areas. Results of the River Wien monitoring project encourage a future shift in state of these sections from a pipe to a processor and corridor of passage for resident animal life, hence accommodating demands by the Water Framework Directive for a ‘good ecological potential’. Unfortunately resistance of society and politicians towards realization of river restoration projects including adequate hydrological connection of fluvial networks remains challenging (Wohl *et al.* 2005; Poole 2010). The implementation of standards for the assessment of restoration success on a national level to expedite scientific knowledge transfer into political response with the aim to improve restoration approaches is recommended (Feminella & Walsh 2005; Palmer *et al.* 2005). Concluding, limnological ecosystems and their resources are of disproportionate particular concern to the environment and socio-economy – fluvial services need an up-scaling as they are important at the global scale (Downing 2009; Booth *et al*. 2016).

REFERENCES

Argerich A., Martí E., Sabater F., Haggerty R. & Ribot M. (2011) Influence of transient storage on stream nutrient uptake based on substrata manipulation. *Aquatic Sciences* 73:365–376.

Alexander R.B., Böhlke J.K., Boyer E.W., David M.B., Harvey J.W., Mulholland P.J., Seitzinger S.P., Tobias C.R., Tonitto C. & Wollheim W.M. (2009) Dynamic modeling of nitrogen losses in river networks unravels the coupled effects of hydrological and biogeochemical processes. *Biogeochemistry* 93:91–116.

Alexander R.B., Smith R.A. & Schwarz G.E. (2000) Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. *Nature* 403:758–761.

Allan J.D. (1995) Stream Ecology. Structure and function of running waters. Chapter 13: Nutrient dynamics. Chapman & Hall, pp. 283–303.

Allan J.D. & Castillo M.M. (2007) Stream Ecology. Structure and function of running waters. Chapter 11: Nutrient dynamics. 2nd edition. Springer, pp. 255–284.

Arango C.P. & Tank J.L. (2008) Land use influences the spatiotemporal controls on nitrification and denitrification in headwater streams. *Journal of the North American Benthological Society* 27:90–107.

Ashkenas L.R., Johnson S.L., Gregory S.V., Tank J.L. & Wollheim W.M. (2004) A stable isotope tracer study of nitrogen uptake and transformation in an old-growth forest stream. *Ecology* 85:1725–1739.

Aschauer A., Zieritz I., Wimmer R., Deutsch K. & Chovanec A. (2006) WGEV Datenband Fliessgewässer, Berichtsteil Donau, vom Kamp bis zur Leitha (ohne March); Moldau. Umweltbundesamt, BMLFUW, Referat VII/1a, 30 pp.

Battin T.J. (2000) Hydrodynamics is a major determinant of streambed biofilm activity: from the sediment to the reach scale. *Limnology and Oceonography* 45(6):1308–1319.

Battin T.J., Kaplan L.A., Newbold J.D. & Hansen C.M.E. (2003) Contributions of microbial biofilms to ecosystem processes in stream mesocosms. Nature 426:439–442.

Bencala K.E. & Walters R.A. (1983) Simulation of solute transport in a mountain pool-and-riffle stream: a transient storage model. *Water Resources Research* 19:718–724.

Bernhardt E.S., Likens G.E., Hall R.O., Buso D.C., Fisher S.G., Burton T.M., Meyer J.L., McDowell W.H., Mayer M.S., Bowden W.B., Findlay S.E.G., MacNeale K.H., Stelzer R.S. & Lowe W.H. (2005a) Can't see the forest for the stream? In-stream processing and terrestrial nitrogen exports. *BioScience* 55:219–230.

Bernhardt E.S., Palmer M.A., Allan J.D., Alexander G., Barnas K., Brooks S., Carr J., Clayton S., Dahm C., Follstad-Shah J., Galat D., Gloss S., Goodwin P., Hart D., Hassett B., Jenkinson R., Katz S., Kondolf G.M., Lake P.S., Lave R., Meyer J.L., O'Donnell T.K., Pagano L., Powell B. & Sudduth E. (2005b) Synthesizing US river restoration efforts. *Science* 308*(*5722*):*636-637.

Booth D.B., Roy A.H., Smith B. & Capps K.A. (2016) Global perspectives on the urban stream syndrome. *Freshwater Science* 35(1):000–000; doi:10.1086/684940

Bottacin-Busolin A., Singer G., Zaramella M., Battin T.J. & Marion A. (2009) Effects of streambed morphology and biofilm growth on the transient storage of solutes. *Environmental Science & Technology* 43:7337–7342.

Boulton A.J. (1999) An overview of river health assessment: philosophies, practice, problems and prognosis. *Freshwater Biology* 41:469–471.

Bukaveckas P.A. (2007) Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. *Environmental Science & Technology.* 41:1570–1576.

Bunn S.E., Davies P.M. & Mosisch T.D. (1999) Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biology* 41:333-345.

Burns D.A. (1998) Retention of NO_3^- in an upland stream environment: a mass balance approach. *Biogeochemistry* 40:73–96.

Butturini A. & Sabater F. (1998) Ammonium and phosphate retention in a Mediterranean stream: hydrological versus temperature control. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1938–1945.

Butturini A. & Sabater F. (1999) Importance of transient storage zones for ammonium and phosphate retention in a sandy-bottom Mediterranean stream. *Freshwater Biology* 41:593–603.

Comte J. & del Giorgio P.A. (2010) Linking the patterns of change in composition and function in bacterioplankton successions along environmental gradients. *Ecology* 91:1466–1476.

Cooper A.B. & Cooke J.G. (1984) Nitrate loss and transformation in two vegetated headwater streams. *New Zealand Journal of Marine and Freshwater Research* 18:441–450.

Craig L.S., Palmer M.A., Richardson D.C., Filoso S., Bernhardt E.S., Bledsoe B.P., Doyle M.W., Groffman P.M., Hassett B.A., Kaushal S.S., Mayer P.M., Smith S.M. & Wilcock P.R. (2008) Stream restoration strategies for reducing river nitrogen loads. *Frontiers in Ecology and Environment* 6(10):529–538.

D'Angelo D.J., Webster J.R., Gregory S.V. & Meyer J.L. (1993) Transient storage in Appalachian and Cascade mountain streams as related to hydraulic characteristics. *Journal of the North American Benthological Society* 12:247–255.

Dahm N.C., Grimm N.B., Marmonier P., Vallet H.M. & Vervier P. (1998) Nutrient dynamics at the interface between surface waters and groundwaters. *Freshwater Biology* 40:427–451.

Decamps H. (1996) The renewal of floodplain forests along rivers: a landscape perspective. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* 26(1):35–59.

DIN EN ISO 9963-1 (1996) Wasserbeschaffenheit - Bestimmung der Alkalinität - Teil 1: Bestimmung der gesamten und der zusammengesetzten Alkalinität (ISO 9963-1:1994) Deutsche Fassung EN ISO 9963-1:1995; Water quality - determination of alkalinity - part 1: determination of total and composite alkalinity (ISO 9963-1:1994), German version EN ISO 9963-1:1995.

DIN EN ISO 9963-2 (1996) Wasserbeschaffenheit - Bestimmung der Alkalinität - Teil 2: Bestimmung der Carbonatalkalinität (ISO 9963-2:1994), Deutsche Fassung EN ISO 9963-2:1995; Water quality - determination of alkalinity - part 2: determination of carbonate alkalinity (ISO 9963-2:1994), German version EN ISO 9963-2:1995.

DIN EN 1189:1996-12 (1996) Wasserbeschaffenheit - Bestimmung von Phosphor - Photometrisches Verfahren mittels Ammoniummolybdat; Water quality - determination of phosphorus - ammonium molybdat spectrometric method.

DIN EN 26777:1993-04 (1993) Wasserbeschaffenheit - Bestimmung von Nitrit - Spektrometrisches Verfahren (ISO 6777:1984); Water quality - determination of nitrite - molecular absorption spectrometric method.

DIN 38405-29:1994-11 (1994) Deutsche Einheitsverfahren zur Wasser-, Abwasser- und Schlammuntersuchung - Anionen (Gruppe D) - Teil 29: Photometrische Bestimmung von Nitrat mit Sulfosalizylsäure (D 29), ISO 7890-3:1988, modifiziert; German standard methods for the examination of water, waste water and sludge - anions (group D) - part 29: spectrometric determination of nitrate with sulfosalicylic acid (D 29), ISO 7890-3:1988, modified.

Dodds W.K., Jones J.R. & Welch E.B. (1998) Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. *Water Research* 32:1455–1462.

Dodds W.K., López A.J., Bowden W.B., Gregory S., Grimm N.B, Hamilton S.K., Hershey A.E., Martí E., McDowell W.H., Meyer J.L., Morrall D., Mulholland P.J, Peterson B.J., Tank J.L., Valett H.M., Webster J.R. & Wollheim W. (2002) Nitrogen uptake as a function of concentration in streams. *Journal of the North American Benthological Society* 21(2):206–220.

Doyle M.W., Stanley E.H. & Harbor J.M. (2003) Hydrogeomorphic controls on phosphorus retention in streams. *Water Resource Research* 39(6):1147; doi:10.1029/2003WR002038

Downing J.A. (2009) Plenary lecture global limnology: up-scaling aquatic services and processes to the planet earth. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* 30(8):1149–1166.

Dudgeon D., Arthington A.H., Gessner M.O., Kawabata Z., Knowler D.J., Lévêque C., Naiman R.J., Prieur-Richard A.H., Soto D., Stiassny M.L. & Sullivan C.A. (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* 81:163–182.

Ensign S.H. & Doyle M.W. (2005) In-channel transient storage and associated nutrient retention: evidence from experimental manipulations. *Limnology and Oceanography* 50(6):1740–1751.

Ensign S.H. & Doyle M.W. (2006) Nutrient spiraling in streams and river networks. *Journal of Geophysical Research* 111, G04009; doi:10.1029/2005JG000114

Feminella J.W. & Walsh C.J. (2005) Urbanization and stream ecology: an introduction to the series. *Journal of the North American Benthological Society* 24(3):585–587.

Fink M.H., Moog O. & Wimmer R. (2000) Fließgewässer – Naturräume Österreichs. Monographien, Band 128. Wasserwirtschaftskataster, Umweltbundesamt, 110 pp.

Fisher S.G., Gray L.J., Grimm N.B. & Busch D.E. (1982) Temporal succession in a desert stream ecosystem following flash flooding. *Ecological Monographs* 52(1):93–110.

Fisher S.G., Grimm N.B., Martí E., Holmes R.M. & Jones J.Jr.B. (1998) Material spiraling in stream corridors: a telescoping ecosystem model. *Ecosystems* 1:19–34.

Fuchsberger J. (2009) Hyporheischer Wasser- und Nährstoffaustausch im Wienfluss-Retentionsbecken 6. Diploma Thesis. Department of Limnology, University of Vienna, 51 pp.

Gabriel O., Balla D., Kalettka T., Schilling C. & Zessner M. (2006) Estimating the influence of increased nitrate loads from wastewater on phosphorus mobility in river sediments. *Acta Hydrochimica et Hydrobiologica* 34(4):333–345.

Glaser S. (2001) The effects of flood-control reservoirs in the Wien River: applying bioreactors to study dissolved organic carbon and ammonium dynamics. Diploma Thesis. Department of Limnology, University of Vienna, 43 pp.

Gore J.A. (1996) Discharge measurements and streamflow analysis. In: Hauer F.R. & Lamberti G.A. (Eds), Methods in Stream Ecology. Academic Press London, pp. 53–74.

Gooseff M.N., LaNier J., Haggerty R. & Kokkeler K. (2005) Determining in-channel (dead zone) transient storage by comparing solute transport in a bedrock channel–alluvial channel sequence, Oregon, USA. *Water Resources Research* 41, W06014; doi:10.1029/2004WR003513

Gooseff M.N., Hall R.O.Jr. & Tank J.L. (2007) Relating transient storage to channel complexity in streams of varying land use in Jackson Hole, Wyoming. *Water Resources Research* 43: W01417; doi:10.1029/2005WR004626

Grimm N.B. (1987) Nitrogen dynamics during succession in a desert stream. *Ecology* 68:1157–1170.

Grimm N.B., Sheibley R.W., Crenshaw C.L., Dahm C.N., Roach W.J. & Zeglin L.H. (2005) N retention and transformation in urban streams. *Journal of the North American Benthological Society* 24:626–642.

Groffman P.M., Bain D.J., Band L.E., Belt K.T., Brush G.S., Grove J.M., Pouyat R.V, Yesilonis I.C. & Zipperer W.C. (2003) Down by the riverside: urban riparian ecology. *Frontiers in Ecology and Environment* 1(6):315–321.

Groffman P.M., Dorsey A.M., & Mayer P.M. (2005) N processing within geomorphic structures in urban streams. *Journal of the North American Benthological Society* 24:613–625.

Gücker B. & Boëchat I.G. (2004) Stream morphology controls ammonium retention in tropical headwaters. *Ecology* 85(10):2818–2827.

Hagedorn F., Schleppi P., Waldner P., Flühler H. (2000) Export of dissolved organic carbon and nitrogen from gleysol dominated catchments – the significance of water flow paths. *Biogeochemistry* 50:137–161.

Hager U. (2002) Post-restoration nutrient and organic matter dynamics in an urban stream and its adjacent wetlands. Diploma Thesis. Department of Limnology, University of Vienna, 58 pp.

Hall, R.Jr.O. (2003) A stream's role in watershed nutrient export. *Proceedings of the National Academy of Science of the United States of America* 100(18):10137–10138.

Hall R.Jr.O., Bernhardt E.S. & Likens G.E. (2002) Relating nutrient uptake with transient storage in forested mountain streams. *Limnology and Oceanography* 47(1):255–265.

Hall R.Jr.O. & Tank J.L. (2003) Ecosystem metabolism controls nitrogen uptake in streams in Grand Teton National Park, Wyoming. *Limnology and Oceanography* 48(3):1120–1128.

Hanafi S., Grace M.R. & Hart B.T. (2006) Can nutrient spiralling be used to detect seasonal nutrient uptake in a foresed steam? *Water, Air, and Soil Pollution: Focus* 6:403–411.

Hart D.R. (1995) Parameter estimation and stochastic interpretation of the transient storage model for solute transport in streams. *Water Resources Research* 31:323–328.

Harvey J.W. & Wagner B.J. (2000) Quantifying hydrologic interactions between streams and their subsurface hyporheic zones. In: Jones J.A. & Mulholland P.J., Streams and Ground Waters. Academic Press New York, pp. 3–44.

Harvey J.W., Conklin M.H. & Koelsch R.S. (2003) Predicting changes in hydrologic retention in an evolving semi-arid alluvial stream. *Advances in Water Resources*, 26:939–950.

Hein T. (2001) Hydrochemie, Schwebstoffdynamik und organische Pools unter Berücksichtigung der Funktion des Hochwasserschutzes in den Wienfluss Retentionsbecken – Jahresbericht 2001 und Zusammenfassung 1999 bis 2001. Bundesministerium f. Land- u. Forstwirtschaft Umwelt u. Wasserwirtschaft, 42 pp.

Hein T. (2002a) Restrukturierung der Retentionsbecken: Bedeutung für den Nährstoffhaushalt und die Selbstreinigungskapazität des Wienflusses. Neues Leben am Wienfluss. *Perspektiven, Heft 1/2*, pp. 18–25.

Hein (2002b) Hydrochemie, Schwebstoffdynamik und organische Pools unter Berücksichtigung der Funktion des Hochwasserschutzes in den Wienfluss Retentionsbecken: Einfluss der neugestalteten Wienfluss-Becken (Becken VI, VII) auf die Nährstoffdynamik des Wienflusses. Jahresbericht 2002. Bundesministerium f. Land- u. Forstwirtschaft Umwelt u. Wasserwirtschaft, 30 pp.

Hein T., Reckendorfer W., Thorp J.H. & Schiemer F. (2005) The role of slackwater areas for biogeochemical processes in rehabilitated river corridors: examples from the Danube. *Archiv für Hydrobiologie Suppl: Large Rivers* 15(1–4):425–442.

Hoellein T.J., Tank J.L., Rosi-Marshall E.J. & Entrekin S.A. (2009) Temporal variation in substratum-specific rates of N uptake and metabolism and their relative contribution at the reach scale. *Journal of the North American Benthological Society*. 28(2): 305-318.

Hölting B. & Coldewey W.G. (2009) Hydrogeologie. Einführung in die allgemeine und angewandte Hydrogeologie. Kapitel 3.2: Hohlräume im Untergrund. 7. Auflage. Spektrum Akademischer Verlag, pp. 11–21.

Howarth R.W., Billen G., Swaney D., Townsend A., Jaworski N., Lajtha K., Downing J.A., Elmgren R., Caraco N., Jordan T., Berendse F., Freney J., Kudeyarov V., Murdoch P. & Zhao-Liang Z. (1996) Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: natural and human influences. *Biogeochemistry* 35:75–139.

Illies J. (1978) Limnofauna Europeae. 2. Auflage. G. Fischer Verlag, Stuttgart, New York; Swets & Zeitlinger B.V., Amsterdam.

Jungwirth M., Muhar S. & Schmutz S. (2002) Re-establishing and assessing ecological integrity in riverine landscapes. *Freshwater Biology* 47:867–887.

Katzmann M. & Forster R. (2002) Nach 100 Jahren fließt der Wienfluss wieder in seinem alten Bett. Neues Leben am Wienfluss. *Perspektiven, Heft 1/2*, pp. 26–34.

Keckeis H. (2002) Die Rückkehr des Flusses: neue Habitate für bedrohte Flussfische in der Stadt. Neues Leben am Wienfluss. *Perspektiven, Heft 1/2*, pp. 49–54.

Keckeis H., Fesl C., Hoyer H., Schludermann E., Scheder C., Forster R. & Katzmann M. (2008) River Wien restoration project: improvement of the ecological condition of a heavily modified river in an urban environment. In: Proceeding of the 4th ECRR Conference on River Restoration, Italy, Venice S. Servolol Island. Chapter 7, Session 6, pp. 571–582.

Kim B.K.A., Jackman A.P. & Triska F.J. (1992) Modeling biotic uptake by periphyton and transient storage of nitrate in a natural stream. *Water Resources Research* 28:2743–2752.

Kitchell J.F., O'Neill R.V., Webb D., Gallepp G.W., Bartell S.M., Koonce J.F. & Ausmus B.S. (1979) Consumer regulation of nutrient cycling. BioScience 29:28–34.

Kjeldahl J. (1883) Neue Methode zur Bestimmung des Stickstoffs in organischen Körpern. *Fresenius Zeitschrift für Analytische Chemie*, S. 366–382.

Klocker C.A., Kaushal S.S., Groffman P.M., Mayer P.M. & Morgan R.P. (2009) Nitrogen uptake and denitrification in restored and unrestored streams in urban Maryland, USA. *Aquatic Sciences* 71(4):411–424.

Konecny R., Aschauer A., Chovanec A., Wimmer R. & Keckeis H. (2002) Heavily Modified Waters in Europe – Case Study Wienfluss. Report to the: Federal Environment Agency, Vienna, 22 pp.

Kondolf G.M., Lisle T.E. & Wolman G.M. (2003) Bed Sediment Measurement. In: Kondolf G.M. & Piégay H.J. (Eds), Chapter 13: Tools in Fluvial Geomorphology. Wiley & Sons Ltd., pp. 347–395.

Kondolf G.M. & Wolman G.M. (1993) The sizes of salmonid spawning gravels. Water Resources Research 29:2275–2285.

Krumbein W.C. & Pettijohn F.J. (1938) Manual of Sedimentary Petrography. New York: Appleton-Century, Inc., pp. 31–33.

Küffer H., Richterich R., Kraft R., Peheim E. & Colombo J.P. (1975) Die Bestimmung des Chlorids in Plasma und Serum (Quecksilber [II]-Thiocyanat-Methode) mit dem Greiner Electronic Selective Analyzer GSA II. *Zeitschrift f. Klinische Chemie u. Klinische Biochemie* 13. Jahrgang, Heft 5:203–211.

Ladinig G., Lazowski W. & Zuckerstätter C. (1999) Der neue Wienfluss – Natur und Technik im Einklang. Wissen – Service für eine grossartige Stadt, Magistrat der Stadt Wien – MA 45, 34 pp.

Lautz L.K. & Siegel D.I. (2007) The effect of transient storage on nitrate uptake lengths in streams: an inter-site comparison. *Hydrological Processes* 21:3533–3548.

Leitner K. (2016) The efficiency of in-stream ammonium uptake along a gradient of ammonium loading. Master Thesis. Department of Water, Atmosphere and Environment, University of Natural Resources and Life Sciences Vienna, 77 pp.

Lewis W.M.Jr, Melack J.M., McDowell W.H., McClain M., Richey J.E. (1999) Nitrogen yields from undisturbed watersheds in the Americas. *Biogeochemistry* 46:149–162.

Loreau M. (2008) Biodiversity and ecosystem functioning: the mystery of the deep sea. *Current Biology* 18(3):126–128.

Martí E. & Sabater F. (1996) High Variability in temporal and spatial nutrient retention in Mediterranean streams. *Ecology* 77(3):854–869.

Martin L.A., Mulholland P.J., Webster J.R. & Valett H.M. (2001) Denitrification potential in sediments of headwater streams in the southern Appalachian Mountains, USA. *Journal of the North American Benthological Society* 20(4):505–519.

McClain M.E., Boyer E.W., Dent C.L., Gergel S.E., Grimm N.B., Groffman P.M., Hart S.C., Harvey J.W., Johnston C.A., Mayorga E., McDowell W.H. & Pinay G. (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems* 6:301–312.

Meyer J.L. & Likens G.E. (1979) Transport and transformation of phosphorus in a forest stream ecosystem. *Ecology* 60:1255–1269.

Meyer J.L., McDowell W.H., Bott T.L., Elwood J.W., Ishizaki C., Melack J.M., Peckarsky B.L., Peterson B.J. & Rublee P.A. (1988) Elemental dynamics in streams. *Journal of the North American Benthological Society* 7:410–432.

Meyer J.L., Paul M.J. & Taulbee W.K. (2005) Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society* 24(3):602-612.

Minshall G.W., Petersen R.C., Cummins K.W., Bott T.L., Sedell J.R., Cushing C.E. & Vannote R.L. (1983) Interbiome comparison of stream ecosystem dynamics. *Ecological Monographs* 53:1–25.

Mitsch W.J. & Gosselink J.G. (2015) Wetlands. Chapter 6: Wetland biogeochemistry. 5th edition. Wiley & Sons, pp. 179–215.

Moog O. (1990) Makrobenthologische Aspekte bei der Wiederherstellung naturnaher Flußabschnitte. *Wiener Mitteilungen* 88:55–103.

Mulholland P.J. (1992) Regulation of nutrient concentrations in a temperate forest stream: roles of upland, riparian, and instream processes. *Limnology and Oceanography* 37(7):1512–1526.

Mulholland P.J., Helton A.M., Poole G.C., Hall R.Jr.O., Hamilton S.K., Peterson B.J., Tank J.L., Ashkenas L.R., Cooper L.W., Dahm C.N., Dodds W.K., Findlay S., Gregory S.V., Grimm N.B., Johnson S.L., McDowell W.H., Meyer J.L., Valett H.M., Webster J.R., Arango C., Beaulieu J.J., Bernot M.J., Burgin A.J., Crenshaw C., Johnson L., Merriam J., Niederlehner B.R., O'Brien J.M., Potter J.D., Sheibley R.W., Sobota D.J. & Thomas S.M. (2008) Stream denitrification across biomes and its response to anthropogenic nitrate loading. *Nature* 452:202–205.

Mulholland P.J., Marzolf E.R., Webster J.R., Hart D.R. & Hendricks S.P. (1997) Evidence that hyporheic zones increase heterotrophic metabolism and phosphorus uptake in forest streams. *Limnology and Oceanography* 42(3):443–451.

Mulholland P.J., Steinmann A.D. & Elwood J.W. (1990) Measurement of phosphorus uptake length in streams: comparisons of radiotracer and stable PO_4 releases. *Canadian Journal of fisheries and Aquatic Sciences* 47(12):2351–2357.

Mulholland P.J. & Webster J.R. (2010) Nutrient dynamics in streams and the role of *J-NABS*. *Journal of the North American Benthological Society* 29(1):100–117.

Munn N.L. & Meyer J.L. (1990) Habitat-specific solute retention in two small streams: an intersite comparison. *Ecology* 71(6):2069–2082.

Murphy J. & Riley J.P. (1962) A single method for the determination of phosphate in natural waters. *Analytica Chimica Acta*, 27:31–36.

Newbold J.D. (1992) Cycles and spirals of nutrients. In: Calow P. & Petts G.E. (Eds), The rivers handbook, Volume 1. Blackwell Scientific, Oxford, England, pp. 379–408.

Newbold J.D., Bott T.L., Kaplan L.A., Dow C.L., Jackson J.K., Aufdenkampe A.K., Martin L.A., Van Horn D.J. & De Long A.A. (2006) Uptake of nutrients and organic C in streams in New York City drinking-water-supply watersheds. *Journal of the North American Benthological Society* 25(4):998–1017.

Newbold J.D., Elwood J.W., O'Neill R.V. & Sheldon A.L. (1983a) Phosphorus dynamics in a woodland stream ecosystem: a study of nutrient spiralling. *Ecology* 64(5):1249–1265.

Newbold J.D., Elwood J.W., O'Neill R.V. & Van Winkle W. (1981) Measuring nutrient spiralling in streams. *Canadian Journal of Fisheries and Aquatic Sciences* 38(7):860–863.

Newbold J.D., Elwood J.W., Schulze M.S., Stark R.W. & Barmeier J.C. (1983b) Continuous ammonium enrichment of a woodland stream: uptake kinetics, leaf decomposition, and nitrification. *Freshwater Biology* 13:193–204.

Newbold J.D., O'Neill R.V., Elwood J.W. & Van Winkle W. (1982) Nutrient spiralling in streams: implications for nutrient limitation and invertebrate activity. *The American Naturalist* 120:628–652.

Newcomer-Johnson T.A., Kaushal S.S., Mayer P.M., Smith R.M. & Sivirichi G.M. (2016) Nutrient retention in restored streams and rivers: a global review and synthesis. *Water* 8, 116; doi:10.3390/w8040116

Oberhofer A. (2002) Neugestaltung des Wienflusses, Bereich Aufhof – Landschaftsplanung. Neues Leben am Wienfluss. *Perspektiven, Heft 1/2*, pp. 14–15.

O'Brien J.M. & Dodds W.K. (2008) Ammonium uptake and mineralization in prairie streams: chamber incubation and short-term nutrient addition experiments. *Freshwater Biology* 53:102–112.

O'Brien J.M., Dodds W.K., Wilson K.C., Murdock J.N., & Eichmiller J. (2007) The saturation of N cycling in Central Plains streams: ^{15}N experiments across a broad gradient of nitrate concentrations. *Biogeochemistry* 84:31–49.

OENORM ISO 7150-1:1987-12-01 (1987) Wasseruntersuchung - Bestimmung von Ammonium - manuelle spektrophotometrische Methode; Water quality - determiantion of ammonium - manual spectrometric method.

Palmer M., Bernhardt E., Chornesky E., Collins S., Dobson A., Duke C., Gold B., Jacobson R., Kingsland S., Kranz R., Mappin M., Martinez M.L., Micheli F., Morse J., Pace M., Pascual M., Palumbi S., Reichman O.J., Simons A., Townsend A. & Turner M. (2004) Ecology for a crowded planet. *Science* 304:1251-1252.

Palmer M., Allan J.D., Meyer J., Bernhardt E. S. (2007). River restoration in the twenty-first century: data and experiential knowledge to inform future efforts. *Restortation Ecology* 15(3):472–481.

Palmer M. (2008) http://palmerlab.umd.edu/restoration_course_docs/2008resources/Nutrient_Processing.pdf, pp. 1–4.

Palmer M.A., Bernhardt E.S., Allan J.D., Lake P.S., Alexander G., Brooks S., J. Carr, Clayton S., Dahm C. N. , Follstad Shah J., Galat D.L., Loss S.G. , Goodwin P., Hart D.D., Hassett B., Jenkinson R., Kondolf G.M. , Lave R., Meyer J.L., O'Donnell T.K., Pagano L. & Sudduth E. (2005) Standards for ecologically successful river restoration. *Journal of Applied Ecology* 42:208–217.

Parsons T.R., Maita Y. & Lalli C.M. (1984) A Manual of Chemical and Biological Methods for Seawater Analysis. Chapter 4: Determination of chlorophylls and total carotenoids: spectrophotometric method. Pergamon Press, Oxford, pp. 101–104.

Paul M.J. & Meyer J.L. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics* 32:333–365.

Paul M.J. & Hall R.O. (2002) Particle transport and transient storage along a steam size gradient in the Hubbard Brook Experiment Forest. *Journal of the North American Benthological Society* 21(2):195–205.

Peterson B.J., Wollheim W.M., Mulholland P.J., Webster J.R., Meyer J.L., Tank J.L., Martí E., Bowden W.B., Valett H.M., Hershey A.E., McDowell W.H., Dodds W.K., Hamilton S.K., Gregory S., Morrall D.D. (2001) Control of nitrogen export from watersheds by headwater streams. *Science* 292:86–90.

Pickett S.T.A, Cadenasso M.L., Grove J.M., Nilon C.H., Pouyat R.V., Zipperer W.C. & Costanza R. (2001) Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32:127–157.

Poff N.L., Allan J.D., Bain M.B., Karr J.R., Prestegaard K.L., Richter B.D., Sparks R.E. & Stromberg J.C. (1997) The natural flow regime: a paradigm for river conservation and restoration. *BioScience* 47:769–784.

Poole G.C. (2010) Stream hydrogeomorphology as a physical science basis for advances in stream ecology. *Journal of the North American Benthological Society* 29(1):12–25.

Pusch M., Fiebig D., Brettar I., Eisenmann H., Ellis B.K., Kaplan L.A., Lock M.A., Naegeli M.W. & Traunspurger W. (1998) The role of micro-organisms in the ecological connectivity of running waters. *Freshwater Biology* 40:453–495.

Raab R. (2002) Libellen als Bioindikatoren zur Überprüfung der Effizienz von Revitalisierungsmassnahmen an Wienfluss und Mauerbach. Neues Leben am Wienfluss. *Perspektiven, Heft 1/2*, pp. 55–62.

Richey J.S., McDowell W.H., Likens G.E. (1985) Nitrogen transformations in a small mountain stream. *Hydrobiologia* 124:129–139.

Roberts B.J. & Mulholland P.J. (2007) In-stream biotic control on nutrient biogeochemistry in a forested headwater stream, West Fork of Walker Branch. *Journal of Geophysical Research* 112: G04002; doi:10.1029/2007JG000422

Roberts B.J., Mulholland P.J. & Houser J.N. (2007) Effects of upland disturbance and in-stream restorations on hydrodynamics and ammonium uptake in headwater streams. *Journal of the North American Benthological Society* 26:38–53.

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

Runkel R.L. (2002) A new metric for determining the importance of transient storage. *Journal of the North American Benthological Society* 21(4):529–543.

Sabater F., Butturini A. & Martí E, Muñoz I., Romaní A., Wray J. & Sabater S. (2000) The effects of riparian vegetation removal on nutrient retention in a Mediterranean stream. *Journal of the North American Benthological Society* 19(4):609–620.

Sabater S., Guasch H., Romani A. & Muñoz I. (2002) The effect of biological factors on the efficiency of river biofilms in improving water quality. *Hydrobiologia* 469:149-156.

Sand-Jensen K., Borg D. & Jeppesen E. (1989) Biomass and oxygen dynamics of the epiphyte community in a Danish lowland stream. *Freshwater Biology* 22:431–443.

Seitzinger S.P., Styles R.V., Boyer E.W, Alexander R.B., Billen G., Howarth R.W., Mayer B. & Van Breemen N. (2002) Nitrogen retention in rivers: model development and application to watersheds in the northeastern U.S.A. *Biogeochemistry* 57/58:199–237.

Seitzinger S., Harrison A., Böhlke J.K., Bouwman A.F., Lowrance R., Peterson B., Tobias C. & Van Drecht G. (2006) Denitrification across landscapes and waterscapes: a synthesis. *Ecological Applications* 16:2064–2090.

Simon K.S., Townsend C.R., Biggs B.J.F. & Bowden W.B. (2005) Temporal variation of N and P uptake in 2 New Zealand streams. *Journal of the North American Benthological Society* 24(1):1–18.

Solórzano L. (1969) Determination of ammonia in natural waters by the phenolhypoclorite method. *Limnology and Oceanography* 14: 799-801.

Stofleth J.M., Shields F.Jr. & Fox G.A. (2008) Hyporheic and total storage exchange in small, sand-bed streams. *Hydrological Processes* 22:1885–1894.

Stream Solute Workshop (1990) Concepts and methods for assessing solute dynamics in stream ecosystems. *Journal of the North American Benthological Society* 9(2):95–119.

STREAMES: Stream Reach Management, an Expert System (2001) *Protocols WP2 & WP3*, 79 pp.

Sullivan S.M.P., Watzin M.C. & Hession W.C. (2004) Understanding stream geomorphic state in relation to ecological integrity: evidence using habitat assessments and macroinvertebrates. *Environmental Management* 34(5):669–683.

Tabacchi E., Lambs L., Guilloy H., Planty-Tabacchi A.M., Muller E. & Decamps H. (2000) Impacts of riparian vegetation on hydrological processes. *Hydrological Processes* 14(16-17):2959–2976.

Thackston E.L. & Schnelle K.B.Jr. (1970) Predicting effects of dead zones on stream mixing. *Journal of the Sanitary Engineering Division* 96(SA2):319–331.

Thomas S.A., Valett H.M., Webster J.R. & Mulholland P.J. (2003) A regression approach to estimating reactive solute uptake in advective and transient storage zones of stream ecosystems. *Advances in Water Ressources* 26:965–976.

Trask P.D. (1932) Origin and environment of source sediments of petroleum. Houston: Gulf Publications & Co. (as cited by Inman 1952).

Valett H.M., Crenshaw C.L. & Wagner P.F. (2002) Stream nutrient uptake, forest succession and biochemical theory. *Ecology* 83(10):2888–2901.

Valett H.M., Morrice J.A., Dahm C.N. & Campana M.E. (1996) Parent lithology, surface-groundwater exchange, and nitrate retention in headwater streams. *Limnology and Oceanography* 41(2):333–345.

Vollenweider R.A. & Kerekes J. (1982) In: Eutrophication of waters. Monitoring, assessment and control. OECD Cooperative programme on monitoring of inland waters (Eutrophication control), Environment Directorate, Paris. 154 pp.

Wallace J.B., Webster J.R. & Woodall W.R. (1977) The role of filter feeders in flowing waters. *Archiv für Hydrobiologie* 79:506–532.

Walsh C.J., Roy A.H., Feminella J.W., Cottingham P.D, Groffman P.M. & Morgan R.P. (2005) The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society* 24(3):706–723.

Wassergüte in Österreich Jahresbericht (2010) Perspektiven für Umwelt & Gesellschaft. Umweltbundesamt. Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft/Sektion VII/Nationale Wasserwirtschaft. 141 pp., plus Annex.

Webster J.R. (1975) Analysis of potassium and calcium dynamics in stream ecosystems on three Southern Appalachian watersheds of contrasting vegetation. PhD dissertation. University of Georgia, Athens, U.S.A.

Webster J.R. & Patten B.C. (1979) Effects of watershed perturbation on stream potassium and calcium dynamics. *Ecological Monographs* 49:51–72.

Web[1], www.wien.gv.at/umwelt/gewaesser/hochwasserschutz/wienfluss

Weigelhofer G., Zornig H. & Hein T. (2005) Hydrochemie, Schwebstoffdynamik und organische Pools unter Berücksichtigung der Funktion des Hochwasserschutzes in den Wienfluss Retentionsbecken: Einfluss der neugestalteten Wienfluss-Becken (Becken VI, VII) auf die Nährstoffdynamik des Wienflusses. Jahresbericht 2005. Bundesministerium f. Land- u. Forstwirtschaft, Umwelt u. Wasserwirtschaft, 37 pp.

Weigelhofer G., Welti N. & Hein T. (2013) Limitations of stream restoration for nitrogen retention in agricultural headwater streams. *Ecological Engineering* 60:224–234.

Wimmer R. & Moog O. (1994) Katalog der Flussordungszahlen österreichischer Fließgewässer. Monographien, Band 51, Umweltbundesamt, 581 pp.

Wohl E., Angermeier P.L, Bledsoe B., Kondolf G.M., MacDonnell L., Merritt D.M., Palmer M.A., Poff N.L. & Tarboton D. (2005) River restoration. *Water Resources Research* 41:1–12, W10301, doi:10.1029/2005WR003985

PICTURE DOCUMENTATION

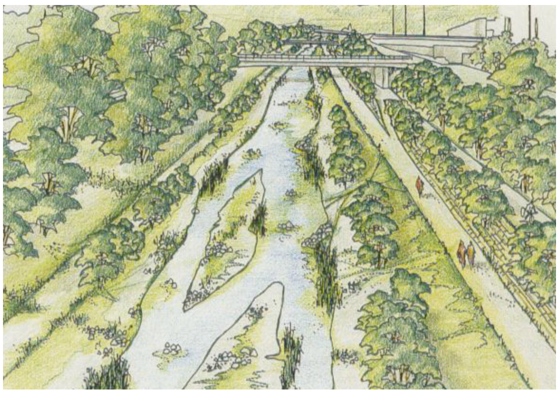
Origin of pictures was noted when traceable. Much to the authors regret not in every case the picture right of ownership was possible to descry. In the case of copyright infringement please notify the author at your earliest convenience. Pictures 7 and 11–16 were taken by the author herself and are free off copyright.

Pic. 5 Coloured copperplate engraving by Johann Friedrich Wizani (1822) showing an idealized view from the left River Wien banks to St. Charles church; © Wien Museum.

Pic. 6 Former public bathing-place at sand deposited banks of the already constrained River Wien in the early 20^{th} century; © not traceable (printed in: Ladinig *et al.* 1999, p. 27).

Pic. 7 River Wien upstream of the Kennedy-bridge (Vienna, Hietzing) during average discharge.

Pic. 8 River Wien upstream of the Kennedy-bridge during the flood of 1997 (HQ100; © Institute of Soil Bioengineering and Landscape Construction, University of Natural Resources and Applied Life Sciences, Vienna.



Pic. 9 Drawing of a possible restoration scenario for the River Wien within Vienna; © not traceable (printed in: Ladinig *et al.* 1999, p. 33).

Pic. 10 Natural upper flow section of stream Mauerbach (in the Vienna Woods; reference for River Wien; © not traceable (printed in: Konecny *et al.* 2002, p. 10).

Pic. 11 A weir-shutter in opened position at the flood-control reservoirs.

Pic. 12 Below solute addition study restructured reach II, showing the recently restored river-bed in 2002.

Pic. 13 Above reach II during elevated discharge. River Wien flows through the reed.

Pic. 14 View of solute addition study restructured reach I upper part.

Pic. 15 Conflux of River Wien and stream Mauerbach.

Pic. 16 Above channelized reach III showing sediment deposits at the left bank.

GERMAN SUMMARY / ZUSAMMENFASSUNG

Einfluss von Restaurierungsmaßnahmen auf den hydrologischen Gesamtrückhalt und die biogeochemische Nährstoffretention eines urbanen Flusses: eine experimentelle Nährstoffzugabe-Studie am Wienfluss (Wien, Österreich)

Der Einfluss von Restaurierungsmaßnahmen der Flussbettstruktur auf den hydrologischen Gesamtrückhalt und die biogeochemische Nährstoffrückhaltekapazität des Wienflusses vierter Ordnung am westlichen Stadtrand von Wien (Österreich) wurde untersucht. Das bessere Verständnis interner Fluss-Strukturen und zugehöriger Prozesse zweier Fließstrecken unterschiedlichen Restaurierungsgrades sowie ökologischen Zustandes war der Focus dieser Studie. Die gewonnen Einblicke ergänzen die vorliegenden Ergebnisse des Monitoringprojektes "Hydrochemie, Schwebstoffdynamik und organische Pools unter Berücksichtigung der Funktion des Hochwasserschutzes in den Wienfluss-Retentionsbecken". Für diese Zielsetzung wurde eine experimentelle Nährstoffzugabestudie bei Niederwasser durchgeführt. Drei Einmalzugaben einer Phosphat (als Na_2HPO_4) - Ammonium (als NH_4Cl) - Natriumchlorid ($NaCl$) Lösung in das Oberflächenwasser wurde im Juli / August / September 2002 umgesetzt. Daten des konservativen Tracers und der biochemisch-reaktiven Nährstoffe flossen in ein eindimensionales Transport-Rückhaltemodell ein. Zusätzlich wurde die Studie auf einen hartverbauten Flussabschnitt ausgeweitet, um potentielle Chancen für den Hochwasserschutz und die Nährstoffdynamik des Wienflusses zu erfassen.

Die Versuchsstrecke II, ein reaktiviertes Feuchtgebiet gelegen in den seit der Revitalisierung in das Abflussgeschehen integrierten Hochwasser-Retentionsbecken, wies die höchste geomorphologische Heterogenität sowie hydrologischen Gesamtrückhalt auf. Die errechnete Flussbett-Komplexität (relative Summe der Sinuosität, Varianz der Gewässerbreite wie auch -tiefe und des Sortierungskoeffizients der Sedimentzusammensetzung) erreichte das zweifache von Versuchsstrecke I, einer revitalizierten riffle-pool Sequenz (Untiefen-Kolk-Abfolge), und das vierfache von Versuchsstrecke III, eines kanalisierten Laufes. Vor allem die Varianz der Flussbreite wie -tiefe, als auch der Anteil an Feinsediment, kontrollierte den hydrologischen Gesamtrückhalt. Die geschaffenen strukturellen Bedingungen resultierten sowohl in einer ausgeprägten Verlangsamung der Fließgeschwindigkeit als auch in einem Anstieg der hydrologischen Aufenthaltszeiten. 34 – 51 % der errechneten Querschnittsfläche des Durchflusses wurden als transiente Speicherzonen (*TSZ*, Kompartimente welchen den longitudinalen flussabwärts gerichteten Wassertransport verzögern) identifiziert. In der Versuchsstrecke II verweilte das Wasser in der *TSZ* 1.6 und 2.2 Mal länger verglichen mit den Versuchsstrecken I und III. Zudem trug die hydrologische Verweildauer in der *TSZ* stärker zum Gesamtrückhalt bei als das Ausmaß der *TSZ*.

Die Aufnahmekapazitäten sowie -geschwindigkeiten für Phosphat ($P\text{-}PO_4$) waren in der Versuchsstrecke I tendenziell größer als in der Versuchsstrecke II. Der hydrologische Gesamtrückhalt der restrukturierten Versuchsstrecken schien von geringer Wichtigkeit für die $P\text{-}PO_4$-Rückhaltekapazität. Gleichermaßen wurden keine Korrelationen von Retentionsparametern mit geomorphologischen Faktoren gefunden, obwohl Sedimentdaten andeuteten, dass $P\text{-}PO_4$ größtenteils chemisch an Feinsediment adsorbierte. Massenbilanzen zeigten, dass mehr Stickstoff als Phosphor in den revitalisierten Fließgewässerabschnitten zurückgehalten wurden. In der Versuchsstrecke III, im Gegensatz, hatte der hydrologische Gesamtrückhalt Einfluss auf die $P\text{-}PO_4$-Aufnahmekapazitäten. S_W (Nährstoffaufnahmelänge) korrelierte mit A_S/A (Verhältnis zwischen der Querschnittsfläche der freifließenden Wassersäule und der Querschnittsfläche der *TSZ*), als auch mit der Größe der *TSZ*.

Die Restaurierung physikalischer Flussbettcharakteristika resultierte in einer Verbesserung der strukturellen und funktionalen Eigenschaften des Wienflusses. Erworbene Resulte, insbesonders bezüglich der Kopplung zwischen Geomorphologie und hydrologischem Gesamtrückhalt, bieten weitere Informationen zur Rehabilitation des verbleibenden hartverbauten Flussabschnitts als auch für zukünftige Prioritäten im Wasserbau.