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RIPARIAN BUFFER STRIPS AS A MULTIFUNCTIONAL TOOL IN AGRICULTURAL LANDSCAPES

Effects of Riparian Forest Buffers on In-Stream Nutrient Retention in Agricultural Catchments

Gabriele Weigelhofer,* Jennifer Fuchsberger, Bernadette Teufl, Nina Welti, and Thomas Hein

In northeastern Austria, marshlands have been turned into the most productive arable land of the country. As a result, most headwater streams show structurally degraded channels, lacking riparian buffer zones, which are heavily loaded with nutrients from the surrounding crop fields. The present study examines whether longitudinally restricted riparian forest buffers can enhance the in-stream nutrient retention in nutrient-enriched headwater streams. We estimated nutrient uptake from pairwise, short-term addition experiments with $^{15}NH_4$, NH_4, PO_4, and NaCl within reaches with riparian forest buffers (RFB) and degraded reaches (DEG) of the same streams. Riparian forest buffers originated from the conservation of the pristine vegetation or from restoration measures. Hydrologic retention was calculated with the model OTIS-P on the basis of conductivity breakthrough curves from the salt injections. A significant increase in surface transient storage was revealed in pristine and restored RFB reaches compared with DEG reaches due to the longitudinal step-pool pattern and the frequent occurrence of woody debris on the channel bed. Ammonium uptake lengths were significantly shorter in RFB reaches than in DEG reaches, resulting from the higher hydrologic retention. Uptake velocities did not differ significantly between RFB and DEG reaches, indicating that riparian forest buffers did not affect the biochemical nutrient demand. Uptake of $^{15}NH_4$ was mainly driven by autotrophs. Net PO_4 uptake was not affected by riparian forest buffers. The study shows that the physical and biogeochemical effects of riparian forest buffers on the in-stream nutrient retention are limited in the case of highly eutrophic streams.

One of the greatest challenges in stream restoration is the mitigation of downstream nutrient transport from intensely used agricultural catchments. The establishment of riparian buffer zones along stream margins can be an effective tool in reducing diffuse inputs from adjacent crop fields (e.g., Parkyn and Davies-Colley, 2003; Sabater et al., 2003; Mayer et al., 2005; Craig et al., 2008). However, buffer design, soil characteristics, climate, and groundwater flow paths may significantly affect buffer effectiveness (Mayer et al., 2005; Verstraeten et al., 2006). Drainage pipes and ditches, which are common in agricultural landscapes, bypass riparian buffers and transport nutrients directly into the stream (Leeds-Harrison et al., 1999). Longitudinal discontinuities in the buffer zones may reduce the riparian retention of nutrients further (Leeds-Harrison et al., 1999).

In Austria, riparian forest buffers are preferred to grass buffers because they induce a more natural and stable stream morphology (Sweeney et al., 2004). Apart from their ability to retain nutrients from entering the stream ecosystem (*riparian nutrient retention*), riparian forest buffers may enhance the capacity of the stream to retain nutrients dissolved in the water column (*in-stream nutrient retention*) (e.g., Hall et al., 2002; Webster et al., 2003; Craig et al., 2008). Flow obstructions, like debris dams and submerged roots, increase the benthic surface area as well as the water travel time, thereby shortening nutrient spiraling lengths (Gücker and Boëchat, 2004; Ensign and Doyle, 2005; Bukaveckas, 2007). Leaves and branches from the riparian vegetation may serve as additional C source for the heterotrophic community, thereby increasing the biological demand for N and P (Roberts et al., 2007; Aldridge et al., 2009; Klocker et al., 2009).

In Austria, the restoration of riparian forest buffers at stream margins is often spatially restricted due to land ownership, land use, and geomorphological situations, resulting in a patchwork of restored and degraded reaches along the stream course. Due to these discontinuities in the buffer zones, the direct input of particulate and dissolved matter from adjacent crop fields into

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*Corresponding author (gabriele.weigelhofer@wkl.ac.at).
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5585 Guilford Rd., Madison, WI 53711 USA

G. Weigelhofer, J. Fuchsberger, B. Teufl, N. Welti, and T. Hein, Univ. of Natural Resources and Life Sciences, Inst. of Hydrobiology and Aquatic Ecosystem Management, Vienna, Austria; G. Weigelhofer, N. Welti, and T. Hein, WasserCluster Lunz, A-3293 Lunz/See, Austria; B. Teufl, Univ. of Vienna, Dep. of Limnology, Vienna, Austria. Assigned to Associate Editor Wim Chardon.

Abbreviations: As, transient storage zone area; CPOM, coarse particulate organic matter; DEG, degraded reaches without riparian forest buffers; Hb, Herbertsbrunn stream; Hip, Hipples stream; RFB, reaches with riparian forest buffers; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Sw, uptake length; U, uptake rate; Vf, mass transfer coefficient.

the streams is high in agricultural catchments (Weigelhofer et al., 2008). The present study examines whether spatially restricted riparian forest buffers can enhance the in-stream nutrient retention in nutrient-enriched headwater streams, thus compensating for deficits in riparian nutrient retention. To address this question, we selected two restored and two morphological pristine reaches with riparian forest buffers and compared them with degraded sections within the same streams regarding hydrologic retention and nutrient uptake. We tested the following hypotheses: (i) In reaches with riparian forest buffers, flow obstructions on the channel bed will lead to a decrease in whole-reach current velocity and an increase in residence time, thereby leading to an increase in hydrologic retention; (ii) due to the additional C supply, reaches with riparian forest buffers will show an increase in biological nutrient demand of the benthic compartments; and (iii) the higher hydrologic retention and the higher nutrient demand will result in an increased in-stream nutrient uptake in reaches with riparian forest buffers compared with degraded reaches.

Materials and Methods

Study Area

The northeastern part of Austria, known as the Weinviertel, is characterized by intensive agriculture of wheat, barley, and wine, among others (Weigelhofer et al., 2008). As part of the Northern Vienna Basin catchment, it consists of gravel, sand, and clay sediments with highly fertile soils. In the 18th and 19th centuries, many headwater streams were artificially excavated to turn the former marshland into arable land. As a result, most streams show straightened, deeply incised channels characterized by steep, V-shaped banks. Bank vegetation consists primarily of reed, grass, and other herbaceous plants (e.g., *Phragmites* sp., *Urtica dioica* L.). Natural forested riparian zones are scarce in the study area, with crop fields extending to the upper bank margin. Stream sediments consist of sand and silt with high organic matter accumulations and are anoxic below the first centimeters of depth. Due to diffuse inputs from the surrounding crop fields, most headwater streams are heavily loaded with nutrients.

Over the past 10 to 30 yr, haphazard restoration measures have been performed to improve the ecological state of the streams. Restoration measures usually extend over 300 to 1000 m of stream length, along which the stream is re-routed into a meandering channel. On both banks, 5- to 10-m-wide riparian buffer zones are constructed where riparian vegetation is planted [e.g., *Salix sp.* and *Alnus glutinosa* (L.) Gaertn., among others].

For this study, we selected four headwater streams that have adjacent riparian forest buffers over a length of at least 300 m (RFB reaches). A site map showing the locations of the study streams is presented in Supplemental Fig. S1. At the Stronsdorf stream (Str; 16°17′ E, 48°40′ N) and Stuetzenhofen stream (Stu; 16°39′ E, 48°44′ N) sites, the riparian forest buffer originated from a restoration 20 to 30 yr ago (Table 1). At the Hipples stream (Hip; 16°24′ E, 48°30′ N) and Herbertsbrunn stream (Hb; 16°43′ E, 48°40′ N) sites, remnants of the original riparian vegetation are present. Restored and morphological pristine reaches were characterized by a closed riparian canopy and

Table 1. Hydromorphological, hydrologic retention, and nutrient uptake parameters of reaches with and without riparian forest buffers in the Weinviertel, Austria. *n* = 24.

Site†	Condition	Discharge	Water depth	Stream width	Current velocity	NH_4–N	PO_4–P	NO_3–N	*D*§	As/A	α	NH_4 Sw	NH_4 *U*	NH_4 Vf	PO_4 Sw	PO_4 *U*	PO_4 Vf
		L s^{-1}	10^{-2} m	m	m s^{-1}	—— μg L^{-1} ——		mg L^{-1}	m^2 s^{-1}		10^{-4} s^{-1}	10^3 m	μg m^{-2} s^{-1}	10^{-5} m s^{-1}	10^3 m	μg m^{-2} s^{-1}	10^{-5} m s^{-1}
Hb RFB	pristine	0.5 (0.1)‡	6.7 (5.8)	0.7 (0.4)	0.06 (0.2)	54 (10)	7 (1)	1.8 (0.2)	0.06 (0.01)	0.5 (0.1)	0.2 (0.1)	0.3 (0.1)	0.2 (0.0)	0.4 (0.1)	0.1 (0.0)	0.2 (0.1)	1.2 (0.3)
Hb DEG	degraded	2.8 (0.2)	7.7 (0.9)	0.7 (0.3)	0.12 (0.2)	32 (8)	11 (3)	0.9 (0.1)	0.09 (0.01)	0.1 (0.0)	1.3 (0.2)	0.3 (0.0)	0.7 (0.1)	2.3 (0.1)	0.1 (0.2)	0.5 (0.1)	4.6 (0.5)
Hip RFB	pristine	12.0 (11.7)	11.2 (1.0)	1.1 (0.2)	0.10 (0.0)	70 (35)	68 (10)	6.1 (1.1)	0.07 (0.05)	5.0 (9.0)	0.7 (0.5)	0.3 (0.0)	3.2 (1.7)	6.5 (4.1)	1.8 (2.2)	1.2 (1.2)	2.3 (2.5)
Hip DEG	degraded	19.4 (19.6)	8.3 (1.0)	0.6 (0.2)	0.54 (0.4)	72 (5)	74 (19)	4.7 (2.7)	0.39 (0.44)	0.1 (0.0)	1.1 (0.3)	2.8 (3.1)	1.0 (0.1)	1.8 (0.8)	1.2 (0.4)	2.0 (1.7)	2.9 (1.7)
Str RFB	restored	9.6 (12.4)	9.3 (2.1)	0.8 (0.2)	0.07 (0.06)	132 (119)	274 (345)	6.2 (4.0)	0.08 (0.08)	3.1 (6.3)	2.8 (4.2)	0.6 (0.3)	2.0 (1.3)	1.5 (0.9)	3.2 (4.3)	3.7 (4.1)	2.2 (4.1)
Str DEG	degraded	13.1 (11.5)	8.1 (3.0)	1.2 (0.5)	0.08 (0.04)	135 (102)	322 (254)	7.6 (5.5)	0.05 (0.03)	0.1 (0.1)	5.7 (3.7)	0.9 (0.3)	2.5 (1.7)	1.5 (1.3)	1.3 (1.1)	2.8 (1.8)	1.4 (1.1)
Stu RFB	restored	7.3 (4.3)	6.2 (1.2)	0.8 (0.1)	0.14 (0.1)	50 (30)	53 (22)	5.4 (2.2)	0.15 (0.02)	3.6 (5.8)	2.5 (1.7)	0.3 (0.3)	1.5 (1.7)	2.9 (2.2)	1.9 (2.1)	0.5 (0.3)	0.7 (0.4)
Stu DEG	degraded	8.7 (4.8)	9.0 (3.0)	0.7 (0.1)	0.15 (0.1)	40 (15)	55 (22)	4.7 (1.9)	0.13 (0.08)	0.1 (0.1)	5.8 (7.6)	0.8 (0.4)	0.7 (0.1)	1.6 (0.6)	3.2 (1.8)	0.5 (0.07)	1.3 (1.3)

† DEG, degraded reaches without riparian forest buffers; Hb, Herbertsbrunn stream; Hip, Hipples stream; RFB, reaches with riparian forest buffers; Str, Stronsdorf stream; Stu, Stuetzenhofen stream.

‡ Values are means with SD in parentheses.

§ α, transient storage exchange coefficient; As/A, relative extension of transient storage zone; *D*, dispersion coefficient; Sw, nutrient uptake length; *U*, nutrient uptake rate; Vf, mass transfer coefficient.

meandering channels with a more or less distinct step-pool pattern and frequent woody debris accumulations on the channel bed (1–4 per 10 m stream length). Each of these RFB reaches was compared with a degraded section of the same stream situated within a distance of 0.5 to 1.5 km downstream of the RFB reach. These degraded (DEG) reaches exhibited a visually uniform, straightened, and often deeply incised channel lacking woody debris accumulations on the channel bed (Table 1). Due to heavy shading by trees at RFB reaches and reed or terrestrial herbaceous plants at DEG reaches, the RFB and DEG reaches were primarily heterotrophic (unpublished data).

To compare the effects of riparian forest buffers on the hydrologic retention and the in-stream nutrient uptake, we performed consecutive short-term nutrient addition experiments at the RFB and DEG reaches of the same stream. Pairs of reaches were characterized by similar topography, sediment characteristics, discharge, and nutrient background. In total, 14 pairwise nutrient addition experiments were performed at baseflow between summer 2007 and summer 2010 at Str (June 2007, July 2007, July 2009, April 2010, July 2010), Stu (July 2009, October 2009, April 2010, July 2010), Hip (July 2007, July 2010, October 2010), and Hb (May 2009, June 2010).

Study Design

The injection of a reactive solute together with a conservative tracer is a well established method to model solute dynamics in streams (Stream Solute Workshop, 1990). The method gives an insight into hydrologic (e.g., advection, dispersion, transient storage) and biogeochemical (e.g., adsorption, assimilation) processes that determine water and solute transport and retention on the reach scale. In combination with stable isotope additions, underlying mechanisms responsible for the net loss of the respective nutrient can be elucidated.

We performed short-term nutrient additions with NH_4Cl, $Na(H_2PO_4)\cdot 2H_2O$, and NaCl as conservative tracer according to the Stream Solute Workshop (1990) protocol. In short, nutrients and NaCl were injected consecutively into each of the 200-m-long study reaches for 2 to 3 h. We recorded conductivity with an electrical conductivity meter (HQ40d; Hach Lange, Düsseldorf, Germany) at the end of each reach to measure changes in relative salt concentration with time. Before the injection and after salt concentrations reached plateau conditions, we collected water samples at five stations every 40 m downstream of the injection point at each study reach. Nutrient concentrations during plateau conditions were 2 to 4 times the background concentrations. Filtered water samples were analyzed for Cl, NH_4–N, NO_3–N, and PO_4–P concentrations using standard colorimetric methods (APHA, 1998) within 48 h after sampling. At each sampling date and study reach, channel characteristics were measured along five to seven cross-sectional profiles, including water depth and channel width.

We performed stable isotope injections with $^{15}NH_4Cl$ over 5 to 6 h at Str and Stu in April 2010 to study short-term $^{15}NH_4$ uptake by different benthic compartments. Methods have been previously published for the use of $^{15}NH_4$ in stream uptake measurements (Mulholland et al., 2000; Webster et al., 2003). Target enrichment in the water column was 5 atm% ^{15}N. Water sampling followed the same protocol as previously described. Triplicates of sediment and phytobenthos were taken at the five stations of each study reach before and at the end of the addition experiments. Water samples were analyzed for ^{15}N-NH_4 following Lachouani et al. (2010). Sediment samples were dried, ground, and analyzed for C and N concentrations and isotope abundances with an elemental analyzer (EA 1110; CE Instruments, Milan, Italy) connected to an isotope ratio mass spectrometer (DeltaPLUS; Finnigan MAT, Bremen, Germany).

Calculations and Statistics

Nutrient concentrations during plateau conditions were divided by Cl concentrations to correct for dilution by groundwater inputs. We calculated nutrient uptake parameters from the longitudinal decline of the corrected nutrient concentrations during plateau conditions via a first-order uptake regression curve (Stream Solute Workshop, 1990). The nutrient uptake length, Sw (m), is the average travel distance of a dissolved atom before it is taken up. It is calculated as the negative inverse of the slope, k, of the uptake regression curve by Eq. [1]:

$$\mathrm{Sw} = -\frac{1}{k} \qquad [1]$$

The mass transfer coefficient, Vf (m s^{-1}), is the vertical velocity at which a nutrient moves from the water column to the benthic compartment. This parameter is related to uptake length, Sw, via Eq. [2]:

$$\mathrm{Vf} = \frac{d \times v}{\mathrm{Sw}} \qquad [2]$$

where d is the mean water depth (m) and v is the mean current velocity (m s^{-1}). The nutrient uptake rate, U (mg m^{-2} s^{-1}), is the total amount of nutrients that is taken up per unit area of stream bottom and per unit time. It is calculated using Eq. [3]:

$$U = \frac{C \times Q}{\mathrm{Sw} \times w} \qquad [3]$$

where C is the nutrient background concentration (mg L^{-1}), Q is the stream discharge (L s^{-1}), and w is the average stream width (m). The uptake length for $^{15}NH_4$ was calculated analogous to the uptake length for the unlabeled NH_4 based on the longitudinal decline of the background-corrected atm% $^{15}NH_4$ in the water column. Uptake of ^{15}N in the sediment and algae samples was calculated from the difference in the $\delta^{15}N$ before and at the end of the addition.

We estimated transient storage parameters by fitting the one-dimensional solute transport model OTIS-P to the observed conductivity breakthrough curves from the salt injection (Runkel, 1998). Transient storage refers to the temporary detainment of solutes in areas of stagnant or slow-moving water (Runkel and Bencala, 1995). The relative extension of the transient storage zone was calculated as the ratio of the storage zone area, As (m^2), to the cross-sectional area, A, of the stream (m^2) as As/A. The transient storage exchange coefficient, α (s^{-1}), represents the exchange rate between the free-flowing water column and the transient storage zone. The dispersion coefficient, D (m^2 s^{-1}), describes the spreading of a solute due to molecular diffusion and shear stress (Runkel and Bencala, 1995).

We used the time-series data of the conductivity curve to estimate the minimum, mean, and maximum whole-reach current velocities. Discharge was calculated as the product of stream width, water depth, and whole-reach mean velocity. The structural heterogeneity of the channel was represented via the coefficients of variation for water depth and channel width. We used the ratio of water depth to stream width as a descriptor for the ratio of the bottom area to the free-flowing water column.

Data were tested for normality and homogeneity of variance with Kolmogorov Smirnov, Shapiro Wilks, and Levene tests. Depending on sample distribution and variances, we conducted paired-sample *t* tests or Wilcoxon tests to detect differences between pairs of RFB and DEG reaches within the same streams. Relationships among discharge, channel morphology, nutrient background, hydrologic retention, and nutrient uptake were explored using Spearman's rank correlations. All statistical analyses were performed using SPSS 15.0 for Windows (SPSS Inc., Chicago, IL).

Results

In general, RFB reaches were shallower and wider than DEG reaches and thus exhibited lower depth-to-width ratios (Table 1). Maximum, mean, and minimum whole-reach current velocities were usually lower at RFB sites. However, statistically significant differences (*t* test; $p < 0.05$; $n = 10$) were only found between pristine RFB reaches (Hip, Hb) and DEG reaches regarding stream width and whole-reach current velocities. Restored RFB reaches (Str, Stu) did not differ significantly from the respective DEG reaches in any of the hydrologic and morphological parameters (*t* test; $p > 0.05$; $n = 18$). Coefficients of variation for depth and width did not differ significantly between RFB and DEG reaches.

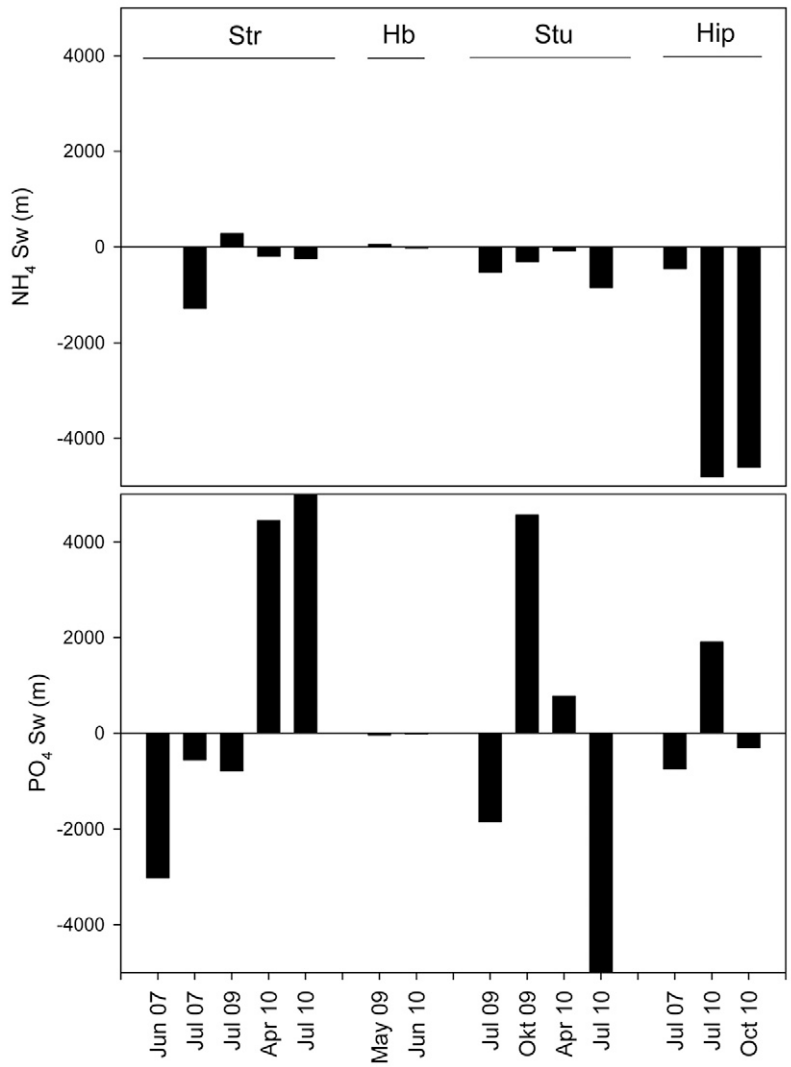

Fig. 1. Differences in NH_4 and PO_4 uptake lengths (Sw) between pairs of stream reaches with and without riparian forest buffers within the same streams ($n = 28$). Negative values indicate a lower uptake length, and positive values indicate a higher uptake length in reaches with riparian forest buffers. Hb, Herbertsbrunn stream; Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream.

Pristine and restored RFB reaches revealed significantly higher As/A ratios than the respective DEG reaches (Wilcoxon-test; $p < 0.05$; $n = 18$). In most cases, RFB reaches showed As/A ratios >1 (Table 1). The As/A ratio was highest in Hip-RFB, which also exhibited the steepest stream gradient and the most distinct step-pool pattern. Dispersion coefficients were significantly lower in pristine RFB reaches vs. DEG reaches (Wilcoxon test; $p < 0.05$; $n = 10$). Transient storage exchange coefficients, α, varied greatly and did not differ significantly between RFB and DEG reaches. In general, α was slightly higher in the DEG reaches (Table 1).

Ammonium uptake lengths were significantly shorter in pristine and in restored RFB reaches compared with the respective DEG sites (*t* test; $p < 0.05$; $n = 9$) (Fig. 1). Reductions in uptake length ranged from 12 to over 90% and were highest at the pristine site Hip-RFB. In only 2 of the 14 pairwise addition experiments were NH_4 uptake lengths longer in the RFB reach (Fig. 1). Ammonium uptake rates differed significantly only between the pristine RFB reaches and the respective DEG reaches (*t* test; $p < 0.05$; $n = 5$). In 10 of the 14 pairwise additions, NH_4 uptake rates were higher in the RFB reaches than in DEG reaches. Differences were most pronounced at Hipples stream, where NH_4 uptake rates at the pristine RFB site were three times those of the DEG site (Table 1). We detected no significant differences in mass transfer coefficients between the RFB and DEG reaches. In general, mass transfer coefficients were slightly higher in the RFB reaches (Table 1).

Phosphate uptake lengths were extremely variable and did not show significant differences between paired sites (*t* test; $p > 0.05$; $n = 9$ (Fig. 1). In 9 of 14 cases, PO_4 uptake lengths were shorter in the RFB reach compared with the respective degraded reach, showing reductions in travel distance of 30 to >95%. On the other hand, in three cases, PO_4 uptake lengths in the RFB reach were 10 to 15 times that of the DEG reach, showing almost no PO_4 uptake. We did not observe any seasonal trends in PO_4 uptake lengths at RFB or DEG reaches (Fig. 1).

Table 2. Correlations among hydromorphological, hydrologic retention, and uptake parameters. Shown are Spearman's rank correlation coefficients and probability levels (n = 24).

	Discharge	As/A	Max. whole-reach current velocity	Mean whole-reach current velocity	Channel width	Water depth	NO_3 background	NH_4 background	PO_4 background
D†	0.56**	ns‡	0.68***	0.73***	ns	ns	ns	ns	ns
As/A	ns		−0.46*	ns	ns	ns	ns	ns	ns
NH_4 Sw	0.50*	−0.67***	0.47*	ns	ns	ns	0.40*	0.42*	0.45*
NH_4 *U*	0.76***	ns	ns	0.49*	0.76***	0.50*	0.61**	0.48*	ns
NH_4 Vf	0.49*	ns	ns	ns	0.50*	ns	ns	ns	ns
PO_4 Sw	ns	ns	ns	ns	ns	ns	ns	ns	ns
PO_4 *U*	ns	ns	ns	ns	ns	ns	ns	0.48*	0.55**
PO_4 Vf	ns	−0.43*	ns	ns	ns	ns	ns	ns	ns

* $p < 0.05$.

** $p < 0.01$.

*** $p < 0.001$.

† As/A, relative extension of transient storage area; *D*, dispersion coefficient; Sw, nutrient uptake length; *U*, nutrient uptake rate; Vf, mass transfer coefficient.

‡ ns, not significant.

Dispersion coefficients were positively correlated with discharge and whole-reach current velocity, whereas As/A decreased with current velocity (Table 2). Ammonium uptake lengths increased significantly with increasing discharge, current velocity, and N background concentrations and decreased with As/A. Ammonium uptake rates and mass transfer coefficients showed positive correlations with discharge and channel width. In addition, NH_4 uptake rates were positively related to nutrient background, water depth, and current velocities. Phosphate uptake rates increased significantly with NH_4 and PO_4 background concentrations, whereas PO_4 mass transfer coefficients decreased with As/A.

Additions of ^{15}N yielded slightly shorter uptake lengths by 5 to 20% than unlabeled NH_4 additions in both streams (Str and Stu). During the isotope additions, $\delta^{15}N$ values increased from 8 to 11‰ in sediments but increased from 10 to almost 30‰ in benthic algae at the RFB reaches (Fig. 2). In the DEG reaches, an increase in $\delta^{15}N$ could only be observed in the phytobenthos of Stu-DEG (from 7 to 13‰).

Discussion

Riparian forest buffers did not induce a significant increase in the cross-sectional channel variability of the investigated study streams. Despite the occurrence of erosion and deposition banks in pristine and restored RFB reaches, water depth, depth-to-width ratios, and coefficients of variation for depth and width lay within the same range in RFB and DEG reaches. Stream width differed significantly only between pristine RFB and DEG reaches. The reason for the mostly subtle differences in cross-sectional variability lies in the extremely small spatial scale of our study streams, where even small clumps of eroded bank material can markedly raise the structural diversity of the channel bed. Similar results were found in a cross-site comparison of 15 morphologically different streams in the Weinviertel in 2007 (Weigelhofer et al., 2008).

Riparian forest buffers affected the hydrologic retention considerably. We observed a significant increase in transient storage in pristine and restored RFB reaches compared with the respective degraded sites. In addition, whole-reach current velocities were significantly lower in pristine RFB reaches. Transient storage can result from hyporheic water exchange and dead-water zones within the active channel (Hall et al., 2002; Grimm et al., 2005). Recent studies have shown that surface transient storage may exceed hyporheic storage in small, sandy bottomed streams (Gücker and Boëchat, 2004; Ensign and Doyle, 2005; Bukaveckas, 2007). The significant increase in transient storage at our RFB sites is probably due to the distinct longitudinal step-pool pattern and the frequent occurrence of woody flow obstructions on the channel bed (Weigelhofer et al., 2008). Studies have shown that additions of coarse woody debris can significantly decrease water velocities and increase hydrologic residence time, thereby increasing surface transient storage (Ensign and Doyle, 2005; Roberts et al., 2007; Klocker et al., 2009).

Because transient storage increases the contact time of the water with the biogeochemical reactive surface, enhanced transient storage is believed to support nutrient uptake

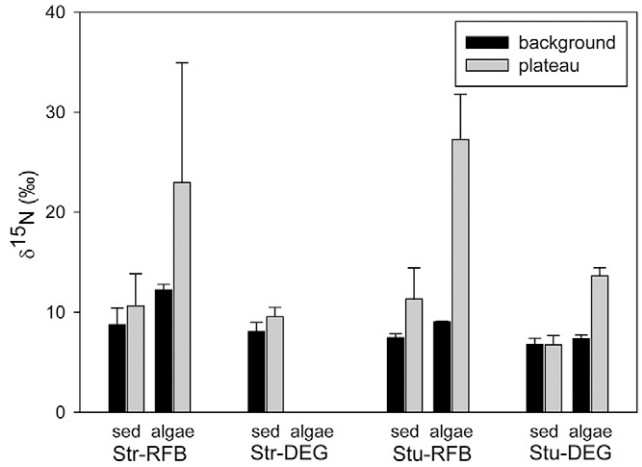

Fig. 2. Changes in the $\delta^{15}N$ (‰) in sediments and benthic algae between background and plateau conditions at reaches with and without riparian forest buffers in Stronsdorf and Stuetzenhofen after short-term $^{15}NH_4$ additions (mean and SD; n = 96). DEG, degraded reach without riparian forest buffer; RFB, reach with riparian forest buffer; sed, sediment samples; Str, Stronsdorf stream; Stu, Stuetzenhofen stream.

(Grimm et al., 2005; Bernhardt and Palmer, 2007). Nutrient uptake length depends on discharge and water velocity (Stream Solute Workshop, 1990; Davis and Minshall, 1999; Hall et al., 2002) and represents the hydrologic component of in-stream nutrient uptake best. Therefore, we expected nutrient uptake lengths to decrease with increasing transient storage and decreasing current velocity (e.g., Hall et al., 2002; Webster et al., 2003), as was the case at RFB reaches. However, this expectation held only for NH_4. Phosphate uptake lengths lacked correlations with hydrologic retention parameters and showed inconsistent patterns between locations and sampling dates. Addition experiments with unlabeled nutrients do not allow following nutrient pathways in detail and can only yield net uptake parameters over the whole stream reach (Stream Solute Workshop, 1990). Net uptake is thus the result of in-stream nutrient uptake minus in-stream nutrient release (Hall et al., 2002; Grimm et al., 2005). Although flow obstructions in the channel increase the contact time with the biologically active surface, they may also induce oxygen-poor conditions at the water–sediment interface, thereby promoting anoxic PO_4 release to the water column (Aldous et al., 2005; Surridge et al., 2007). Mineralization of organic matter may be an additional source of PO_4 in organic-rich sediments (Aldous et al., 2005). Our study streams are characterized by nutrient-rich, anoxic sediments with high organic matter accumulations. Laboratory experiments have revealed a high potential for PO_4 release from these sediments (unpublished data). Therefore, we assume that the high variability in PO_4 uptake and the inconsistent spatial and temporal patterns may be the results of release processes partly overlaying uptake processes.

The mass transfer coefficient represents the benthic demand for nutrients relative to the supply (Hall et al., 2002; Grimm et al., 2005). Our RFB reaches were characterized by increased amounts of coarse particulate organic matter (CPOM) on the channel bed in the form of woody debris and leaf accumulations from the riparian forest buffers. Because the C/N/P ratio of terrestrial CPOM is characterized by a higher C content than that of the benthic microbial community, the decomposition of CPOM requires additional N and P sources, thereby increasing the nutrient demand of the stream ecosystem (Bernot and Dodds, 2005; Aldridge et al., 2009). Therefore, we expected to observe an increase in nutrient uptake velocities in our RFB reaches as a consequence of the increased CPOM availability. In fact, pristine and restored RFB reaches showed slightly higher NH_4 mass transfer coefficients than the respective DEG reaches. However, the differences were small and insignificant, indicating that the additional terrestrial C source had little effect on the activity of the microbial community in these nutrient-enriched streams.

Uptake length is related to uptake rate via nutrient concentrations (see Eq. [3]). According to the Stream Solute Workshop (1990), increasing background concentrations should be outweighed by increasing nutrient uptake rates at unsaturated conditions, leaving nutrient uptake length unaffected. Therefore, positive correlations between nutrient uptake length and background concentrations indicate decreasing uptake efficiencies and a potential saturation of the stream ecosystem. In our study, nutrient uptake length and uptake rates increased with increasing nutrient background concentrations. Together with the rather long uptake lengths, this suggests that our streams are likely approaching saturated conditions, a phenomenon common in streams draining agricultural catchments (e.g., Davis and Minshall, 1999; Bernot and Dodds, 2005; Bernot et al., 2006; Gücker and Pusch, 2006).

Our $^{15}N\text{-}NH_4$ additions indicate that N uptake in our study streams is mainly governed by autotrophs. However, heavy shading restricts the benthic primary production in our study streams during most of the vegetation period and thus may limit autotrophic uptake. On the other hand, anoxic conditions at the water–sediment interface, together with woody debris on the channel bed, may favor denitrification, which can be an important factor for bioreactive N removal in N-enriched streams (Sweeney et al., 2004; O'Brien and Dodds, 2008; Klocker et al., 2009). Further investigations are necessary to clarify the role of autotrophic uptake vs. denitrification in reducing downstream N transport and to elucidate interactions with other nutrient cycles (e.g., phosphate) at the water–sediment interface.

Conclusions

Riparian forest buffers can significantly enhance the surface hydrologic retention and thereby shorten nutrient uptake lengths in nutrient-enriched agricultural streams, even if they are spatially restricted. However, improved in-stream nutrient uptake seems to be rather the result of the higher hydrologic retention than of an increase in the biochemical nutrient demand. Besides, longer travel times may induce the establishment of low oxygen conditions near the water–sediment interface, thereby favoring anoxic processes that may act as sink (e.g., denitrification) or source (e.g., PO_4 release) for nutrients (e.g., Surridge et al., 2007; Klocker et al., 2009). As a consequence, restoration measures that substantially change the channel morphology and the riparian zone may affect the cycling of C, N, and P in different ways and thus may create negative side-effects that threaten the success of the restoration measure (Craig et al., 2008). Our study reveals that in-stream nutrient retention cannot compensate for deficits in riparian nutrient retention when the nutrient supply exceeds the demand significantly (Gücker and Pusch, 2006). To mitigate downstream nutrient transport, the efficiency of riparian nutrient retention must be enhanced by carefully planning riparian buffer design and avoiding anthropogenic by-passes (Verstraeten et al., 2006). In addition, stream restoration should be accompanied by changes in agricultural practices to reduce soil erosion and nutrient loading already at the catchment scale (Gücker and Pusch, 2006; Verstraeten et al., 2006; Craig et al., 2008).

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References

Aldridge, K.T., J.D. Brookes, and G.G. Ganf. 2009. Rehabilitation of stream ecosystem functions through the reintroduction of coarse particulate organic matter. Restor. Ecol. 17:97–106. doi:10.1111/j.1526-100X.2007.00338.x

Aldous, A., P. McCormick, C. Ferguson, S. Graham, and C. Craft. 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restor. Ecol. 13:341–347. doi:10.1111/j.1526-100X.2005.00043.x

APHA. 1998. Standard methods for the examination of water and wastewater. 20th ed. American Public Health Association, Washington, DC.

Bernhardt, E.S., and M.A. Palmer. 2007. Restoring streams in an urbanizing world. Freshwater Biol. 52:738–751. doi:10.1111/j.1365-2427.2006.01718.x

Bernot, M.J., and W.K. Dodds. 2005. Nitrogen retention, removal, and saturation in lotic ecosystems. Ecosystems 8:442–453. doi:10.1007/s10021-003-0143-y

Bernot, M.J., J.L. Tank, T.V. Royer, and M.B. David. 2006. Nutrient uptake in streams draining agricultural catchments of the midwestern United States. Freshwater Biol. 51:499–509. doi:10.1111/j.1365-2427.2006.01508.x

Bukaveckas, P.A. 2007. Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. Environ. Sci. Technol. 41:1570–1576. doi:10.1021/es061618x

Craig, L.S., M.A. Palmer, D.C. Richardson, S. Filoso, E.S. Bernhardt, B.P. Bledsoe, M.W. Doyle, P.M. Groffman, B. Hassett, S.S. Kaushal, P.M. Mayer, S.M. Smith, and P.R. Wilcock. 2008. Stream restoration strategies for reducing river nitrogen loads. Front. Ecol. Environ 6:529–538. doi:10.1890/070080

Davis, J.C., and G.W. Minshall. 1999. Nitrogen and phosphorus uptake in two Idaho (USA) headwater wilderness streams. Oecologia 119:247–255. doi:10.1007/s004420050783

Ensign, S.H., and M.W. Doyle. 2005. In-channel transient storage and associated nutrient retention: Evidence from experimental manipulations. Limnol. Oceanogr. 50:1740–1751. doi:10.4319/lo.2005.50.6.1740

Grimm, N.B., R.W. Sheibley, C.L. Crenshaw, C.N. Dahm, W.J. Roach, and L.H. Zeglin. 2005. N retention and transformation in urban streams. J. North Am. Benthol. Soc. 24:626–642.

Gücker, B., and I.G. Boëchat. 2004. Stream morphology controls ammonium retention in tropical headwaters. Ecology 85:2818–2827. doi:10.1890/04-0171

Gücker, B., and M.T. Pusch. 2006. Regulation of nutrient uptake in eutrophic lowland streams. Limnol. Oceanogr. 51:1443–1453. doi:10.4319/lo.2006.51.3.1443

Hall, R.O., E.S. Bernhardt, and G.E. Likens. 2002. Relating nutrient uptake with transient storage in forested mountain streams. Limnol. Oceanogr. 47:255–265. doi:10.4319/lo.2002.47.1.0255

Klocker, C.A., S.S. Kaushal, P.M. Groffman, P.M. Mayer, and R.P. Morgan. 2009. Nitrogen uptake and denitrification in restored and unrestored streams in urban Maryland, USA. Aquat. Sci. 71:411–424. doi:10.1007/s00027-009-0118-y

Lachouani, P., R. Hood-Nowotny, A. Frank, and W. Wanek. 2010. A suite of sensitive chemical methods to determine the $\delta^{15}N$ of ammonium, nitrate and total dissolved N in soil extracts. Rapid Commun. Mass Spectrom. 24:3615–3623. doi:10.1002/rcm.4798

Leeds-Harrison, P.B., J.N. Quinton, M.J. Walker, C.L. Sanders, and T. Harrod. 1999. Grassed buffer strips for the control of nitrate leaching to surface waters in headwater catchments. Ecol. Eng. 12:299–313. doi:10.1016/S0925-8574(98)00075-5

Mayer, P.M., S.K. Reynolds, M.D. McMutchen, and T.J. Canfield. 2005. Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: A review of current science and regulations. EPA/600/R-05/118. USEPA, Office of Research and Development, Washington, DC.

Mulholland, P.J., J.L. Tank, D.M. Sanzone, W.M. Wollheim, B.J. Peterson, J.R. Webster, and J.L. Meyer. 2000. Nitrogen cycling in a forest stream determined by a ^{15}N tracer addition. Ecol. Monogr. 70:471–493.

O'Brien, J.M., and W.K. Dodds. 2008. Ammonium uptake and mineralization in prairie streams: Chamber incubation and short-term nutrient addition experiments. Freshwater Biol. 53:102–112.

Parkyn, S., and R. Davies-Colley. 2003. Riparian management: How well are we doing? Water Atmos. 11:15–17.

Roberts, B.J., P.J. Mulholland, and J.N. Houser. 2007. Effects of upland disturbance and instream restoration on hydrodynamics and ammonium uptake in headwater streams. J. North Am. Benthol. Soc. 26:38–53. doi:10.1899/0887-3593(2007)26[38:EOUDAI]2.0.CO;2

Runkel, R.L. 1998. One dimensional transport with inflow and storage (OTIS): A solute transport model for streams and rivers. U.S. Geological Survey Water-Resources Investigation Report 98-4018. USGS, Reston, VA.

Runkel, R.L., and K.E. Bencala. 1995. Transport of reacting solutes in rivers and streams. p. 137–164. *In* V.P. Singh (ed.) Environmental hydrology. Kluwer Academic, Dordrecht, The Netherlands.

Sabater, S., A. Butturini, J.-C. Clement, T. Burt, D. Dowrick, M. Hefting, V. Maitre, G. Pinay, C. Postolache, M. Rzepecki, and F. Sabater. 2003. Nitrogen removal by riparian buffers along a European climatic gradient: Patterns and factors of variation. Ecosystems 6:20–30. doi:10.1007/s10021-002-0183-8

Stream Solute Workshop. 1990. Concepts and methods for assessing solute dynamics in stream ecosystems. J. North Am. Benthol. Soc. 9:95–119. doi:10.2307/1467445

Surridge, B.W.J., A.L. Heathwaite, and A.J. Baird. 2007. The release of phosphorus to porewater and surface water from river riparian sediments. J. Environ. Qual. 36:1534–1544. doi:10.2134/jeq2006.0490

Sweeney, B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proc. Natl. Acad. Sci. USA 101:14132–14137. doi:10.1073/pnas.0405895101

Verstraeten, G., J. Poesen, K. Gillijns, and G. Govers. 2006. The use of riparian vegetated filter strips to reduce river sediments loads: An overestimated control measure? Hydrol. Processes 20:4259–4267. doi:10.1002/hyp.6155

Webster, J.R., P.J. Mulholland, J.L. Tank, H.M. Valett, W.K. Dodds, B.J. Peterson, W.B. Bowden, C.N. Dahm, S. Findlay, S.V. Gregory, N.B. Grimm, S.K. Hamilton, S.L. Johnson, E. Marti, W.H. McDowell, J.L. Meyer, D.D. Morrall, S.A. Thomas, and W.M. Wollheim. 2003. Factors affecting ammonium uptake in streams: An inter-biome perspective. Freshwater Biol. 48:1329–1352. doi:10.1046/j.1365-2427.2003.01094.x

Weigelhofer, G., M. Makal, S. Muhar, S. Preis, N. Kreuzinger, and T. Hein. 2008. ProFor (Austria): Integrating the effects of stream structure on nutrient loading patterns in management options for heavily modified streams. p. 697–704. *In* B. Gumiero, M. Rinaldi, and B. Fokkens (ed.) Proc. 4th ECRR, Venice, 16–18 Jun. 2008. CIRF, Venice, Italy.