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„Effects of bank and channel morphology on the sediment quality of agricultural first-order streams“

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ZUSAMMENFASSUNG

Im Nordosten Österreichs befindet sich das Weinviertel, welches sich durch intensive landwirtschaftliche Nutzung auszeichnet. Die meisten Fließgewässer, erster und zweiter Flussordnungszahl, wurden begradigt und weisen daher ein V-förmiges Profil mit steilen Ufern auf. Aufgrund der hohen Nährstoffbelastung bei gleichzeitig geringen Wasserführungen, sowie der intensiven Regulierungs- und Drainagetätigkeiten sind diese Bäche stark beeinträchtigt. Weiters wurden diese Gewässer ihrer Gehölzpufferstreifen beraubt, wodurch die Belastung durch Erosion aus der landwirtschaftlichen Umgebung, lokal und sogar flussabwärts der Eintragsstelle, verstärkt wurde. Daher war das Ziel dieser Studie, die Effekte von Ufer- und Bachbettrenaturierung auf die Sedimentqualität von stark nährstoffbelasteten Bächen in landwirtschaftlichem Einzugsgebiet zu untersuchen.

Natürliche und renaturierte, bewaldete Bachabschnitte wurden mit degradierten Bachabschnitten verglichen. Die Untersuchungsgewässer waren der Hipples Bach, der Stützenhofner Graben, der Stronsdorfer Graben, der Herbertsbrunngraben, sowie der Herrnbaumgartner Graben. Die degradierten Bachabschnitte unterschieden sich von den bewaldeten Bachabschnitten durch fehlende Ufervegetation, sowie durch eine homogene Bachbettmorphologie. Untersucht wurden die Korngrößen, der Nährstoffgehalt und der Anteil des organischen Materials im Sediment der verschiedenen Bachabschnitte. Weiters wurde die räumliche und saisonale Variabilität der Sedimentqualität, sowie die Effekte einer bewaldeten Strecke auf flussabwärts gelegene, degradierte Bachabschnitte an drei Fallbeispielen untersucht. Um mögliche erhöhte Einträge nach Düngeraufbringung festzustellen, wurden die Sedimentproben in obere und untere Schichten unterteilt. Als Folge einer Sedimentqualitätsänderung kann man auch eine Reaktion in der benthischen, mikrobiellen Respiration erwarten, einer zentralen Ökosystemfunktion von Fliessgewässern. Die benthische, mikrobielle Respiration wurde als Konzentrationsunterschied im gelösten Sauerstoff in einem Laborversuch ermittelt. Der Respirationsversuch wurde mit Sedimenten des Hipples Baches, des Stützenhofner Grabens und des Stronsdorfer Grabens durchgeführt.

Diese Untersuchung hat gezeigt, dass der Effekt bewaldeter Ufergehölzstreifen auf die Sedimentqualität stark belasteter, landwirtschaftlicher Bäche gering ist. Bezüglich der hydromorphologischen Parameter unterschieden sich die Gewässer mit und ohne Ufergehölzstreifen hauptsächlich in der Anzahl der Strömungshindernisse, sowie in der

Fläche der Stillwasserbereiche. Generell wies das Sediment dieser Bäche sehr geringe Korngrößen, mit 50 µm im Durchschnitt auf. Daher kann man davon ausgehen, dass ein Großteil der Sedimente durch Erosion aus dem direkten Umland eingetragen wurde. Eine Verbesserung in der durchschnittlichen Korngröße, in den bewaldeten Strecken, konnte nicht nachgewiesen werden, allerdings wurde eine größere Korngrößenvariabilität festgestellt. Höchste Sedimentnährstoffkonzentrationen (Totalphosphor: 1476 µg g^{-1} Trockensediment, N-NO_3: 3,7 µg g^{-1} Trockensediment) wurden in den verbreiterten, deradierten Bachstrecken ermittelt, allerdings konnten signifikante Unterschiede zu den bewaldeten Bachabschnitten nur bei den Phosphorfraktionen nachgewiesen werden. Dies ist darauf zurückzuführen, dass das feine Sediment in den verbreiteten, degradierten Schilfstrecken akkumuliert wird und dadurch auch die gemessenen Phosphorkonzentrationen, aufgrund des partikelgebundenen Phosphors, höher lagen. Im Gegensatz dazu wird das feine Sediment in den kanalisierten, degradierten Stecken eher abtransportiert, was durch die größeren Krongrößen und die geringeren Nährstoffkonzentrationen bestätigt wurde. Diese Ergebnisse deuten darauf hin, dass die Sedimentstruktur und –zusammensetzung hauptsächlich eine Funktion des Partikeltransports sowie der Retention des entsprechenden Bachabschnittes darstellt.

Eine Ursache für die geringen Effekte der Ufergehölzstreifen auf die Sedimentqualität, liegt in der Kleinräumigkeit, sowohl in der Breite als auch in der Länge, der bewaldeten Strecken. Weiters zeichnen sich diese Restaurationsstrecken durch ein mangelhaftes Design der Ufergehölzstreifen aus. Vor allem die Lücken in den Ufergehölzstreifen sowie die Umgehung der Gehölzstreifen durch Drainagerohre beeinflusst deren Effektivität und dadurch können diese Puffer der starken Belastung durch die Landwirtschaft im direkten Umfeld nicht standhalten.

Signifikante Verbesserungen in der Sedimentqualität können daher nur durch Maßnahmen auf Ebene des Einzugsgebietes erreicht werden. Zunächst müssen Erosionshotspots und Eintragspfade des erodierten Materials ausfindig gemacht werden. Weiters müssen Strukturen zum Rückhalt der erodierten Erde, wie zum Beispiel Retentionsbecken oder effiziente Gehölzpufferstreifen, geschaffen werden. Besonders Maßnahmen zur Verringerung der Erosion sowie der Nährstoffbelastung in der direkten Umgebung der Bäche sind notwendig. Durch landwirtschaftliche, sogenannte „best management practices“ könnte dies gewährleistet werden. Um effiziente Restauration landwirtschaftlicher Bäche zu sichern, ist Monitoring der Maßnahmen und deren Effekte auf das Bachökosystem notwendig.

Eine leicht veränderte Version dieser Arbeit wurde bereits im Journal „Environmental Science and Pollution Research“ publiziert (Teufl et al., 2012).

ABSTRACT

In the north-east of Austria, characterized by intensive agricultural land use, most low-order streams are straightened and feature steep V-shaped profiles. Due to high nutrient loading and intense regulation and drainage works, most of the low-order streams are heavily impacted there. Further stream degradation deprives streams of buffers against erosion from the agricultural surrounding and may deteriorate the situation locally and even downstream. Therefore the aim of the study was to investigate the effects of bank and channel restoration on the sediment quality of already nutrient-enriched agricultural headwater streams.

Natural and restored forested stream reaches were compared to degraded stream reaches. Degraded stream reaches differed from other reaches by a lack of riparian vegetation and more homogeneous stream morphology. The stream reaches were investigated with respect to sediment grain size, nutrient concentrations and organic matter content. Furthermore, we determined the spatial and seasonal variability of sediment quality and the impact of forested reaches on down-stream reaches in three case studies. Upper and lower sediment layers were compared to identify increased input after manure application. As sediment quality changes, a change in benthic microbial respiration can be expected as a functional response of the stream ecosystem. Therefore we determined benthic microbial respiration via the change in dissolved oxygen concentrations in laboratory assays.

This study shows that the effect of forested riparian buffers on sediment quality of heavily impacted agricultural low-discharge streams is small. Due to spatially limited restoration reaches and their poor design, their effects on nutrient concentrations and the depth profile are limited. Continuous forested riparian buffers, that are not bypassed, could increase sediment quality. However, significant improvements in sediment composition and structure can only be achieved by additional measures at catchment scale to reduce erosion and loading. Therefore, hotspots of erosion and pathways of soil have to be identified. Retention basins and buffer strips have to be applied and agricultural best management practices are vital in the adjacent surrounding of the streams. To ensure an efficient restoration, these measures and their effect on stream ecosystems have to be monitored.

A slightly different version of this thesis has already been published in the journal of Environmental Science and Pollution Research (Teufl et al., 2012).

KEY WORDS

stream restoration, sediment, agriculture, riparian buffer, nutrients, benthic microbial respiration

ABBREVIATIONS

BMP, best management practice; CPOM, coarse particulate organic matter; CV, coefficient of variation; DB, degraded broadened reaches without riparian forest buffers; DI, degraded incised reaches without riparian forest buffers; FN, forested reaches with a natural riparian buffer; FR, forested reaches with a restored riparian buffer; FRB, forested riparian buffer; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream; Hip, Hipples stream; IP, inorganic phosphorous; OM, organic matter; OP, organic phosphorous; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; TOC, total organic carbon; TN, total nitrogen; TP, total phosphorous;

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INTRODUCTION

Streams are systems that intensively interact with their watershed area. Therefore, human activities in the catchment have strong impacts on streams (Mulholland et al., 2008). Especially agriculture is known to be a major source of pollutants when it comes to soil erosion and concurrent nutrient loading via different input pathways (e.g., Walling, 1999; Sharpley et al., 2003; Noe and Hupp, 2005; Bernot et al., 2006). Increased diffuse inputs of nutrient rich soils from the surrounding crop fields lead to an increase in suspended solids and sedimentation, which may affect habitat quality negatively (Wood and Armitage, 1997; Bhattarai et al., 2009). Agricultural streams are usually affected by multiple stressors like sediment loading, nutrient loading, or water abstraction. As stressors interact (Matthaei et al., 2010), the consequences of stressors are often unpredictable on the basis of knowledge of single effects (Townsend et al., 2008).

Riparian buffers are an approved restoration measure to reduce lateral inputs of soil and nutrients to stream systems (e.g., Lowrance et al., 1984; Peterjohn and Correll, 1984; Osborne and Kovacic, 1993; Lowrance et al., 1997; Naiman and Décamps, 1997; Craig et al., 2008; Bhattarai et al., 2009, Montreuil et al., 2010). In Austria, stream restoration concepts include both the reconstruction of meandering stream courses and the rehabilitation of forested riparian buffers (FRBs) in order to raise habitat heterogeneity and retain diffuse inputs from the surrounding crop fields (Weigelhofer et al., 2011a). However, due to land ownership and agricultural land use, FRBs are spatially restricted in both lateral and longitudinal directions and usually stretch out over less than one kilometer of stream length (Weigelhofer et al. 2011a). Besides, drainage pipes and drainage ditches, which are common in agricultural catchments, function as short-cuts, surpassing these FRBs and transporting dissolved and particulate nutrients directly to the stream (Lowrance et al. 1984; Osborne and Kovacic, 1993; Craig et al., 2008). Such discontinuities in FRBs may reduce the retention efficiency of the riparian buffers markedly or even increase nutrient transport to the stream (Bhattarai et al., 2009) and thus, threaten the ecological success of restoration projects. With an increasing awareness of the importance of fluvial systems, the number of restoration efforts is growing, however, the evaluation of restoration projects is hardly performed (Bernhardt et al., 2007). The overall aim of the current study was to find out whether spatially restricted restoration measures may function as an effective measure to improve sediment structure and quality of agricultural streams, thereby soothing the multiple pressures from agricultural land use.

Streams with FRBs, whether restored or of natural origin, usually differ from streams lacking such zones with respect to channel and bank morphology. In Austria, streams featuring FRBs often show meandering stream courses and exhibit diverse channel and flow patterns due to CPOM (coarse particulate organic matter) input on the channel bed (Aldridge et al., 2009), submerged roots, and step-pool sequences (Montgomery et al., 1995). The increased channel heterogeneity and the resulting increase in surface transient storage may enhance the retention and uptake of dissolved inorganic nutrients in the surface water (Craig et al., 2008; Roberts et al., 2007; Weigelhofer et al., 2008; Weigelhofer et al., 2011a). Besides, Roberts et al. (2007) demonstrated that upland disturbance affects nutrient retention negatively, because CPOM is not delivered from the degraded reach upstream any more. Thus upstream disturbance decreases channel heterogeneity downstream, because of lack of woody debris input.

While the effects of channel heterogeneity on dissolved nutrients have been studied (Roberts et al. 2007; Weigelhofer et al. 2011a and b), little is known about the effects of hydromorphology on sediment structure and composition, especially in systems which are loaded with particle bound nutrients from agricultural catchments. Restoration measures inducing flow alterations often focus on the improvement of local habitat quality (Wood and Armitage, 1997), species performance or biotic interactions (e.g. species richness, composition, and reproduction) (Poff and Zimmerman, 2010), but the effects on sediment composition and sedimentary nutrient concentrations are rarely investigated. It has long been noted that particle retention is a function of the number of flow obstructions and the probability of particles being trapped (Young et al., 1978). Furthermore it is known that fine sediment deposit during low-flow conditions, e.g. at channel margins, within macrophyte stands, at debris dams, and at sites where bed topography and channel structures changes (Wood and Armitage, 1999; Nakamura and Swanson, 1993) Due to different flows, an undulatory bed profile develops, yielding in sorting processes of the sediment (Lisle, 1979). Thus, restoration measures, which include the broadening of the channel and the introduction or removal of flow obstructions, should have significant impacts on fine sediment deposition pattern. As channel heterogeneity influences surface water nutrient retention and sediment deposition, we can assume that sedimentary nutrients are also affected.

The north-eastern part of Austria, the Weinviertel, is characterized by intensive agricultural land use. To reclaim land for crop fields, most streams have been deprived of natural FRBs and have been straightened and regulated. The geological underground is composed of loess and loam and the fertile gley soils exhibit a high erosion potential (Heinrich et al., 2004; Leitner, 2010). Consequently, headwater streams receive high inputs of soil from the surrounding crop fields, leading to generally high accumulations of fine nutrient rich sediments and organic matter in the channel (Weigelhofer et al., 2011b).

The key question of the present study was whether an improved stream and bank morphology, though spatially restricted, can influence sediment structure and composition of headwater streams under the constraint of an overall high nutrient and sediment loading from the agricultural catchment. In specific, we wanted to know, if meandering stream reaches with intact FRBs, that provide woody debris and CPOM, will exhibit a more diverse sediment structure, with larger grain sizes, a higher variability in grain sizes and less nutrient and organic matter concentrations, than degraded stream reaches deprived of FRBs. As community respiration is an important indicator for the functioning of stream ecosystems (Fellows et al., 2006) and is furthermore closely linked to sediment quality, we also wanted to know whether sediments from hydromorphological degraded, restored or near-natural reaches will exhibit differing and type-specific benthic microbial respiration rates.

To answer these questions, we compared headwater reaches featuring FRBs with degraded reaches deprived of FRBs in the agriculturally used Weinviertel catchment in the north-east of Austria. In general, FRB reaches were characterized by meandering channels and diverse flow patterns, while degraded reaches showed straightened channels with uniform flow patterns. As we could not distinguish between effects of bank vegetation and channel morphology, all results will apply to both factors.

We investigated hydromorphology, water quality, sediment structure and composition, as well as benthic microbial respiration rates to test the following hypotheses:

(i) FRB reaches will show larger mean grain sizes and a higher variability in grain sizes than degraded reaches due to more diverse channel and flow patterns.

(ii) The sediments of FRB reaches will contain fewer amounts of nutrients than degraded reaches due to a more heterogeneous sediment structure and a better oxygen supply.

(iii) Effects of FRB reaches on sediments will be perceivable even in degraded reaches immediately downstream of FRB reaches.

(iv) FRB reaches will feature lower benthic microbial respiration rates.

MATERIAL AND METHODS

STUDY AREA

The Weinviertel is a highly productive area in the north-east of Austria. As the Weinviertel is part of the Northern Vienna Basin, it is characterized by silt, sand, and clay sediments. It used to be marshland, but as a result of its fertile soils, it was turned into arable land within the last centuries (Leitner, 2010; Weigelhofer et al., 2011a). Consequently, headwater streams were exposed to many stressors. Because of land reclamation and flood protection, the former meandering swamp streams were straightened and excavated, lowering the groundwater table by about 1.5 m. Today, many headwater streams feature degraded channels and banks. The streams usually show straightened, incised channels with steep, V-shaped profiles. Bank vegetation primarily consists of reed, grass, and herbaceous plants (e.g. *Urtica dioica*, *Carex* sp.). Pristine riparian forests are rare in the Weinviertel. Usually, crop fields reach to the upper bank margins. Due to the lack of riparian buffer zones and the high erosion potential of the silty to loamy soils, most streams feature high accumulations of fine sediments on the channel bed and anoxic conditions in the sediments below a few centimeters of depth. To improve the ecological status of these streams, several restoration projects have been carried out within the last decades. Restoration measures included the reconstruction of the original meandering stream course over reaches of 0.3 – 1 km length, and the plantation of 5 m wide forested riparian buffers (FRBs) on both banks (Weigelhofer et al., 2011a; Figure 1).

In order to study sediment characteristics of morphologically different stream reaches, we selected nine 200 m long study reaches within five headwater streams. All sample streams featured low discharge values of 1 to 5 l sec^{-1} and medium discharge values between 10 and 20 l sec^{-1}. Four of those reaches showed FRBs (FRB reaches), the other five were deprived of FRBs (degraded reaches). FRB reaches consisted of both restored and natural reaches. The degraded reaches were divided into incised and broadened channel segments based on the different stream width and channel profile type. The following paragraph describes the four different stream types in detail.

Natural FRB reaches (FN) showed pristine meandering channels with remnants of natural FRBs on both banks and were found at the Hipples stream (Hip, 16°24’ E, 48°30’ N) and the upper Herbertsbrunn stream (Hbb I, 16°42’ E, 48°40’ N). Restored FRB reaches (FR) were characterized by a reconstructed meandering channel with a restored floodplain forest and were studied at the upper Stronsdorf stream (Str I, 16°17' E, 48°39' N) and the upper Stuetzenhofen stream (Stu I, 16°43' E, 48°44' N). Incised degraded reaches (DI) showed straightened, incised channels without FRBs and were found at the lower Stuetzenhofen stream (Stu II, 16°44' E, 48°44' N), the lower Herbertsbrunn stream (Hbb II, 16°43' E, 48°40' N), and the lower Herrnbaumgarten stream (Hbg I, 16°45' E, 48°41' N). Broadened degraded reaches (DB) exhibited a straightened stream course like the incised reaches (DI), but showed a broadened channel and dense reed stocks on both the banks and the channel bed. As degraded reaches were straightened, neither undercut slopes nor slip-off slopes have developed, thus degraded reaches are characterized by rather symmetrical channel profiles (Leitner, 2010). Broadened reaches were studied at the lower Stronsdorf stream (Str II, 16°17' E, 48°40' N) and the upper Herrnbaumgarten stream (Hbg II, 16°46' E, 48°41' N) (Figure 2).

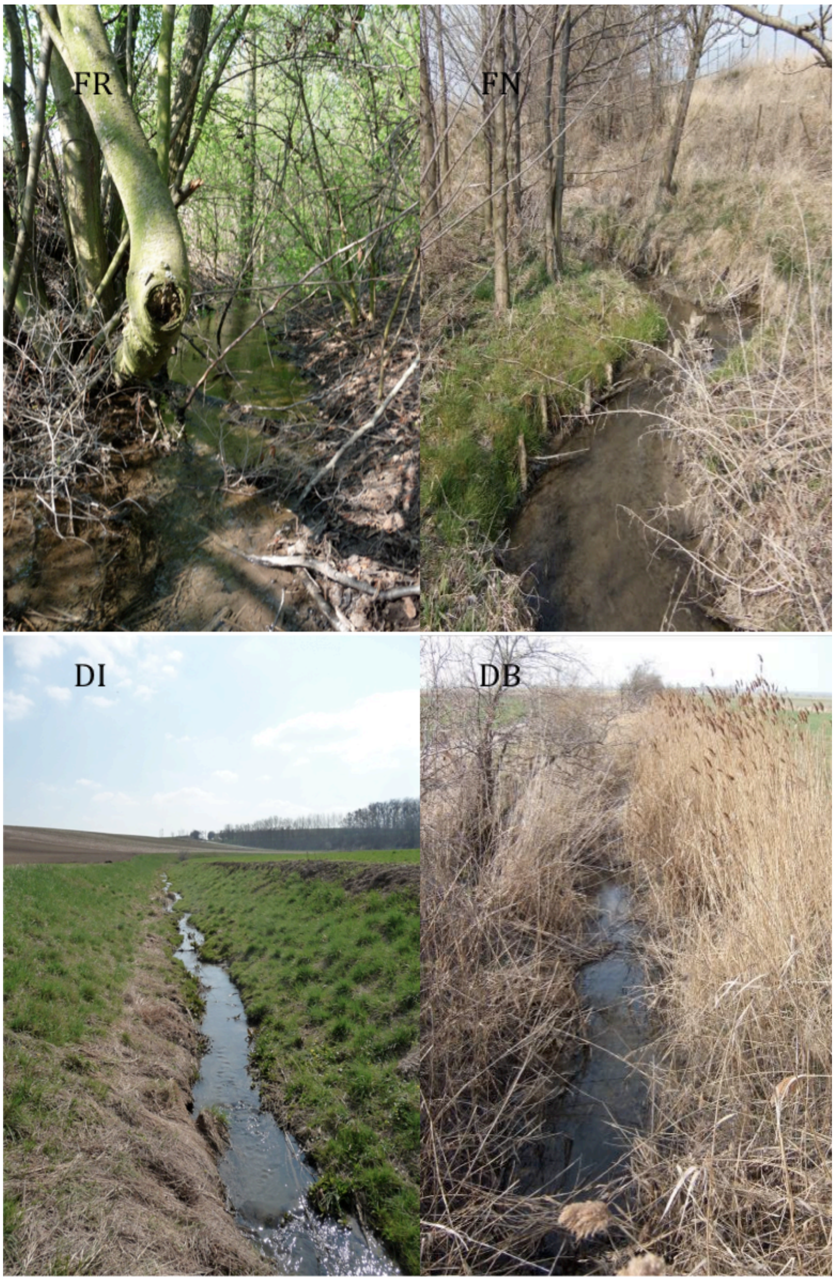

Figure 1. Sample pictures of the 4 morphological stream types. On the top, FRB reaches are shown, whereas degraded reaches are given beneath.
FN, natural FRB reach; FR, restored FRB reach; FRB, forested riparian buffer; DB, broadened degraded reach; DI, incised degraded reach.

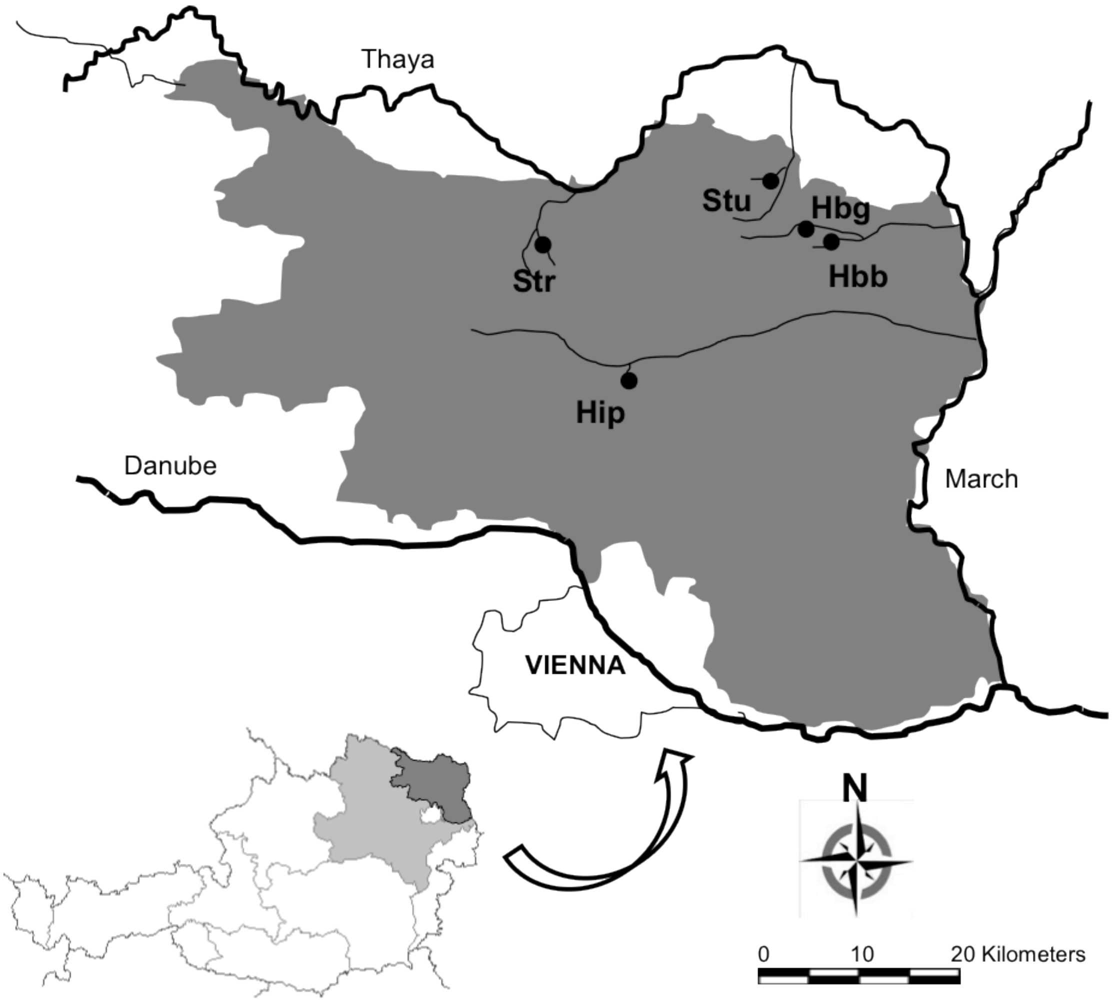

Figure 2: The 5 sampling streams (Hip, Str, Stu, Hbb, and Hbg) are shown within a map of the Weinviertel (Lower Austria, light grey; Weinviertel, dark grey). Large Austrian rivers are also illustrated.
Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.

STUDY DESIGN

In order to study the effects of stream morphology on sediment structure, nutrient concentrations, and organic matter (OM) content in the sediments, we took 3 to 5 sediment samples with a PVC-corer (internal diameter 5 cm) down to a sediment depth of 10 cm at each study reach in April, July, and October 2010. Samples were taken randomly distributed over a 10 m stream section at the end of each reach to keep upstream effects on sediment composition as small as possible. Immediately after taking the samples, the depth of the oxic layer was estimated visually via the coloring of the sediment layers (Rosenberg et al., 2001). To reduce the influence of floods, all samples were taken at least 1 month after flooding.

To investigate differences between the oxygenated sediment surface and deeper layers, the upper 2 cm of the sediment samples of 3 streams (Hip, Stu, and Str) were analyzed separately. For the comparison of sediments among reach types, surface and deeper layers were pooled together. We determined grain size distribution via wet sieving with mesh sizes from 0.0063 mm to 20 mm (ÖNORM, 2002). The OM content was estimated after combustion at 450°C for 4 hours via the ash-free dry weight of the sediment fractions. Inorganic phosphorous and $P\text{-}PO_4$ concentrations in the sediments were determined according to Ruban et al. (2001). Total phosphorous (TP) was measured after acid digestion of the sediments in a microwave oven (CEM MarsXpress) according to USEPA method SW846-3051 (USEPA, 1998). The $N\text{-}NO_3$, $N\text{-}NO_2$, and $N\text{-}NH_4$ concentrations were determined according to the SSSA book series 5 (2005). Total nitrogen (TN) and total organic carbon (TOC) concentrations of the sediments were analyzed after HCL treatment using an elemental analyzer (EA 1110 CE Instruments). The ratio of $\delta^{13}C$ was determined by an isotope ratio mass spectrometer (DeltaPLUS Thermo Finnigan) with an interface (ConFlo III Thermo Finnigan) and two elemental analyzers (Flash EA and EA 1110 CE Instruments). Organic phosphorous and nitrogen concentrations were calculated from the difference between TP or TN and inorganic phosphorous or inorganic nitrogen, respectively.

Water samples were collected at 5 sites along each study reach in monthly to bi-monthly intervals and analyzed for nutrient and DOC concentrations. The $N\text{-}NO_3$, $N\text{-}NO_2$, $N\text{-}NH_4$, and $P\text{-}PO_4$ concentrations were measured colorimetrically (CFA Alliance Instruments, APHA, 1998). The DOC concentration was determined with a TOC analyzer (Sievers 900, GE Analytical Instruments).

At three study streams (Stuetzenhofen stream, Stronsdorf stream, and Hipples stream), we took additional samples along a longitudinal transect to investigate effects of FRB reaches on degraded downstream reaches. For this, we sampled the degraded channel upstream of the FRB reach, the middle and the end of the FRB reach, and approximately 20 m and 200 m downstream of the FRB reach at Stuetzenhofen stream and Stronsdorf stream (in total 5 sites with 6 samples each). Hipples stream had no degraded reach upstream of the FRB reach. Besides, the more distant downstream part was a concrete channel. Therefore, we sampled only the beginning and the end of the FRB reach, and one site approximately 300 m downstream of the FRB reach at this stream.

At 3-5 times during the study period, channel characteristics, including water depth, current velocity (FlowMate), channel width, and bankfull width were measured at low to medium water levels along 5-7 cross-sectional profiles at each study site. Heterogeneity of the channel was expressed as coefficient of variation of water depth, channel width, and current velocity. Once during the study period, the amount of woody debris on the channel bed and the percent coverage of the stream channel by stagnant areas were determined (Laub, 2012).

Additionally, we determined benthic microbial respiration in the sediments of Hipples stream, Stronsdorf stream, and Stuetzenhofen stream to assess gross microbial activity of different sediments under controlled conditions. The incubation experiments were performed in the lab in October 2010, following the method of Hill et al. (2000). Therefore, the first cm of the surface sediment was sampled. 5 g of fresh, wet sediments were incubated in 50 ml stream water within sealed 50 ml centrifuge tubes without headspace. The 50 tubes per sampling site and treatment were kept under continuous shaking in the dark over 36 hours. Tubes with stream water, but without sediments, were treated in the same manner to determine microbial respiration in the water column. In order to study temperature effects on microbial respiration, sediments were incubated at 10 °C, 18 °C, and 26 °C, because these temperatures covered the in-situ temperature range. Oxygen was measured every 4 hours at 5 samples, respectively, to get an oxygen-time-curve. Measurements were conducted with an oxygen meter (HQ40d, Hach Lange). After the experiments, the sediments in the tubes were dried (40 °C, 48 hours) and weighed. Benthic microbial respiration was determined via the decrease in dissolved oxygen concentrations in the tubes after correction against water respiration. We calculated the specific respiration rate by dividing the corrected DO changes in the tubes by the area of the sediments sampled (g O_2 m^{-2} d^{-1}).

STATISTICAL ANALYSES

Normality and homogeneity of variance was tested with Kolmogorov-Smirnov, and Shapiro-Wilk tests. Depending on the distribution, parametric or non-parametric tests were performed. Differences among stream types were tested with One-way-ANOVA and post-hocs tests (Bonferroni, Tamhane) regarding hydromorphology and with Kruskal-Wallis and Mann-Whitney-U tests for sediment characteristics. The Wilcoxon-test was performed to detect differences between sites in water quality and sediment layers. Correlations among hydromorphology, water quality, and sediment parameters were identified by Spearman's or Pearson's correlation. The Friedman test was performed to examine possible differences along the longitudinal transects. To detect differences in benthic microbial respiration between stream types or treatments, Mann-Whitney U-tests were performed. Correlations among benthic microbial respiration and hydromorphological, sediment and water parameters were detected by Spearman's correlation. Statistical analyses were conducted with SPSS 15.0 for Windows.

RESULTS

HYDROMORPHOLOGICAL PARAMETERS

Hydromorphology differed significantly among stream types in water depth, bankfull width, and current velocity (One-way ANOVA, $p<0.01$). However, differences were often small and parameters showed a high variability within groups and reaches. Mean water depth was significantly higher in the broadened degraded reaches (DB) with about 0.15 m compared to the others (0.08-0.09 m; $p < 0.05$, $n = 170$). While stream width did not differ among stream types, bankfull width was significantly larger at DB reaches (1.8 m; $p < 0.01$, $n = 110$). Bankfull width also differed significantly between natural FRB reaches (FN) and degraded incised channels (DI) ($p < 0.01$). Bankfull width was smallest at DI with only 0.9 m and intermediate at FR (1.1 m) and FN (1.3 m). Current velocity differed significantly between FN and DI ($p < 0.01$, $n = 110$). The highest mean current velocity was observed at incised channels (DI, 0.11 m s^{-1}), while natural RFB reaches (FN) showed the smallest current velocity (0.05 m s^{-1}). Only the coefficient of variation of current velocity showed significant differences among stream types (FN-FR: $p = 0.009$; FR-DI: $p = 0.01$, $n = 110$). The number of woody debris based flow obstructions per reach was significantly higher at FRB reaches (FN and FR) than at degraded ones ($p < 0.05$, $n = 52$). Both, natural FRB reaches (FN) and degraded broadened reaches (DB) showed a significant higher percentage of stagnant areas than the others ($p < 0.05$, $n = 140$).

Water depth and stream width correlated positively with bankfull width (Pearson, $p = 0.007$ and 0.0005, respectively, Table 1). The percent coverage of stagnant areas increased with stream and bankfull width and number of woody flow obstructions and decreased with mean current velocity (Pearson, $p = 0.0009$, 0.01, and 0.05, respectively, Table 1).

WATER CHEMISTRY

Water $P\text{-}PO_4$ concentrations featured significant differences between the 2 sampling sites of the Stronsdorf stream (Wilcoxon-test, $p = 0.0004$; $n = 25$) and the Stützenhofen stream (Wilcoxon-test, $p = 0.002$; $n = 27$). The DOC concentration differed significantly at Stronsdorf stream (Wilcoxon-test, $p = 0.0005$; $n = 20$), whereas the Stützenhofen stream sampling sites differed in water $N\text{-}NH_4$ concentrations (Wilcoxon-test, $p = 0.007$; $n = 27$). The DOC concentrations and water $P\text{-}PO_4$ and $N\text{-}NH_4$ concentrations of all samples featured positive correlations amongst each other (Spearman, DOC - $P\text{-}PO_4$: $p < 0.0001$; DOC - $N\text{-}NH_4$: $p = 0.02$; $n = 110$, respectively; $P\text{-}PO_4\text{-}N\text{-}NH_4$: $p < 0.0001$; $n = 239$). $N\text{-}NO_3$ concentrations of the water did not show any correlation to other water chemistry parameter. (Table 2)

Table 1. Correlations among hydromorphological parameters, mean grain size and organic matter content of sediments given as Pearson's rank correlation coefficients and probability levels. (n = 110).
* p< 0.05.
** p < 0.01.
† OM, organic matter; CV, coefficient of variation; n.s., not significant.

	water depth	stream width	bankfull width	flow velocity	flow obstructions	stagnant areas	oxic layer	CV depth	CV width	CV velocity	mean grain size	OM
water depth		**0.35****	**0.27****	n.s.	n.s.	n.s.	n.s.	**-0.21***	n.s.	n.s.	n.s.	n.s.
stream width	**0.35****		**0.83****	n.s.	n.s.	**0.23***	n.s.	**-0.24****	n.s.	n.s.	n.s.	n.s.
bankfull width	**0.27****	**0.83****		n.s.	n.s.	**0.34****	n.s.	**-0.23***	**0.33***	n.s.	n.s.	n.s.
flow velocity	n.s.	n.s.	n.s.		n.s.	**-0.44****	n.s.	**-0.21***	**0.44****	**-0.21***	n.s.	n.s.
flow obstructions	n.s.	n.s.	n.s.	n.s.	n.s.	**0.28***	**0.52****	n.s.	n.s.	n.s.	n.s.	n.s.
stagnant areas	n.s.	**0.23***	**0.34****	**-0.44****	**0.28***		n.s.	n.s.	n.s.	n.s.	n.s.	**0.48****
oxic layer	n.s.	n.s.	n.s.	n.s.	**0.52****	n.s.		n.s.	n.s.	**-0.37****	n.s.	n.s.
CV depth	**-0.21***	**-0.24****	**-0.23***	**-0.21***	n.s.	n.s.	n.s.		n.s.	n.s.	**0.26***	n.s.
CV width	n.s.	n.s.	**0.33***	**0.44****	n.s.	n.s.	n.s.	n.s.		n.s.	**0.81****	**-0.60***
CV velocity	n.s.	n.s.	n.s.	**-0.21***	n.s.	n.s.	**-0.37****	n.s.	n.s.		n.s.	n.s.
mean grain size	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	**0.26***	**0.81****	n.s.		**-0.31****
OM†	n.s.	n.s.	n.s.	n.s.	n.s.	**0.48****	n.s.	n.s.	**-0.60***	n.s.	**-0.31****	

Tabelle 2. Morphological and water chemistry parameters of the sampling sites in the Weinviertel, Austria (means with standard deviation in parentheses). Woody debris based flow obstructions are shown as number per sample reach, stagnant and fast flowing areas in % coverage of the sample reach. The maximum water temperature is shown and the oxygen concentration is given as minimum and maximum. (n = 110)
DOC is given as ppm, n = 158.
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡ FN, forested reaches with a natural riparian buffer; FR, forested reaches with a restored riparian buffer; DB, degraded broadened reaches without riparian forest buffers; DI, degraded incised reaches without riparian forest buffers; DOC, dissolved organic carbon; n.d., not determined; CV, coefficient of variation.

sites †	reach type ‡	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth	CV width	CV velocity	O_2 min.	O_2 max.	DOC	N-NO_3	N-NH_4	P-PO_4
		10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage				mg l^{-1}	mg l^{-1}	ppm	µg l^{-1}	µg l^{-1}	µg l^{-1}
Hip	FN	12	1.03	1.47	0.09	95	48	0.75	0.25	0.77	10.2	10.8	2.13	6787.6	41.2	56.8
		(9)	(0.35)	(0.45)	(0.03)								(0.37)	(113.5)	(11.9)	(11.0)
Str I	FR	9	0.86	1.12	0.08	28	39	0.57	0.22	0.96	8.4	15.9	5.04	6782.1	192.3	78.1
		(3)	(0.18)	(0.18)	(0.06)								(0.59)	(3789.8)	(239.2)	(44.2)
Str II	DB	18	1.35	2.00	0.06	0	67	0.63	0.15	0.96	8.5	11.3	5.89	8589.4	149.7	154.0
		(24)	(0.51)	(0.32)	(0.01)								(0.82)	(5612.6)	(88.9)	(109.3)
Stu I	FR	7	0.81	1.16	0.09	116	50	0.69	0.18	1.08	7.2	15.9	3.43	6190.8	100.9	95.1
		(3)	(0.17)	(0.14)	(0.06)								(0.71)	(6106.7)	(221.3)	(110.4)
Stu II	DI	7	0.74	0.78	0.09	0	34	0.77	0.22	1.16	8.4	10.8	3.90	5276.5	42.4	50.1
		(2)	(0.19)	(0.15)	(0.03)								(1.42)	(3219.1)	(49.7)	(30.7)
Hbb I	FN	7	0.74	1.07	0.04	91	81	0.71	0.18	0.84	8.8	9.3	5.34	2028.1	74.6	149.0
		(4)	(0.23)	(0.19)	(0.02)								(2.75)	(2812.4)	(31.3)	(179.0)
Hbb II	DI	5	054	0.97	0.03	6	0	0.76	0.21	0.90	6.9	9.5	6.75	3033.7	57.6	110.6
		(3)	(0.27)	(0.13)	(0.01)								(1.77)	(3742.1)	(36.2)	(156.7)
Hbg I	DB	10	0.99	1.53	0.10	0	71	0.69	0.51	0.86	9.3	13.0	8.58	4590.5	126.8	457.2
		(2)	(0.67)	(1.21)	(0.03)								(7.88)	(3384.8)	(152.6)	(368.3)
Hbg II	DI	9	0.63	0.83	0.12	7	21	0.70	0.18	0.82	4.4	9.7	6.28	4605.6	114.0	507.6
		(4)	(0.15)	(0.14)	(0.06)								(1.02)	(3563.4)	(246.7)	(355.8)

SEDIMENT COMPOSITION

Mean grain sizes were small and ranged from sand to silt at all sites. The smallest grain sizes with an average of about 37 µm were found at broadened degraded reaches (DB), which differed significantly from all other reaches (Mann-Whitney U-test, $p < 0.05$). FRB reaches (FN and FR) featured a higher variability in grain sizes than the others, showing even proportions of sand (up to 50 %) and small gravel (up to 19 %). Mean grain size were 73 µm and 51 µm, respectively. ($n = 77$, Table 3, Figure 3)

The depth of the oxic layer (Table 3) was significantly higher at FRB reaches (FN and FR) than at degraded reaches (DI and DB) (Mann-Whitney U-test, $p <0.05$, $n = 56$). Especially FN featured oxic layers of more than 10 cm in depth.

Organic matter concentrations (Table 3) were highest at DB and FN (mean values of 7.6 and 7.1 % dry weight, respectively), but only DB differed significantly from FR and DI (Mann-Whitney U-test, $p < 0.01$; $n = 80$). Organic matter content was negatively correlated with mean grain size (Spearman, $p < 0.01$).

All types of stream reaches featured a high spatial and temporal variability in sedimentary nutrient concentrations. The highest nutrient concentrations in the sediments were generally detected at DB (Table 3; Figure 3). Nevertheless, significant differences between DB and the other stream types were only observed regarding TP and inorganic phosphorous concentrations (Kruskal-Wallis test, $p < 0.0001$, $n = 140$). The organic phosphorous fraction of DB did not differ significantly from FN (Mann-Whitney U-test: DB-FR and DB-DI: $p < 0.0001$, $n = 77$ and 79, respectively). Mean TP values of FRB reaches and degraded reaches accounted for 855, 1022, (FN and FR) 914, and 1476 (DI and DB) µg g^{-1} dry sediment, respectively, whereas mean sedimentary SRP values accounted for 1.8, 2.9, 2.2, and 3.7 µg g^{-1} dry sediment, respectively. N-NO_3, N-NO_2, and N-NH_4 showed only small differences among the different stream types. N-NH_4 concentrations were significantly lower at DI (66 µg g^{-1} dry sediment) (Mann-Whitney U-test, DI-FN: $p < 0.0001$; DI-FR: $p = 0.0006$; DI-DB: $p < 0.0001$, $n = 61$, 78, and 77, respectively), whereas N-NO_2 concentrations were lowest at FR (1.0 µg g^{-1} dry sediment) (Mann-Whitney U-test, FR-FN: $p =0.004$; FR-DI: $p = 0.026$; FR-DB: $p = 0.012$, $n = 62$). N-NO_3 concentrations were lowest at DI, however significant differences were only detected for DB (mean values of 1.1 and 3.7 µg g^{-1} dry sediment, respectively) (Mann-Whitney U-test, $p = 0.005$, $n = 77$).

In general, the sediments featured high total phosphorous concentrations of up to 2.7 mg g^{-1} dry sediment and high $N\text{-}NH_4$ concentrations of up to 1.1 mg g^{-1} dry sediment. $N\text{-}NO_3$ and $P\text{-}PO_4$ concentrations in the sediments were comparatively low (mean values of 2.2 and 2.9 µg g^{-1} dry sediment, respectively; Table 3, Figure 3).

The C:N ratio (Table 3) ranged from 4.8 to 13.4 and was lower at FRB reaches (FN and FR; mean values of 7.7 and 8.9, respectively) than at degraded reaches (DI and DB; mean values of 9.6 and 10.1, respectively). FN reaches differed significantly from all other stream types (Mann-Whitney U-test, FN-FR: $p = 0.036$; FN-DI: $p = 0.007$; FN-DB: $p = 0.0004$; $n = 93$), whereas FR featured significant differences only to DB (Mann-Whitney U-test, $p = 0.003$; $n = 93$). No differences were observed in N:P ratios among different stream types. Concerning the $\delta^{13}C$ value, DB differed significantly from FRB reaches (FN and FR; Mann-Whitney U-test, $p < 0.0001$ and $p = 0.008$, respectively; $n = 92$).

The $N\text{-}NO_3$, $N\text{-}NO_2$, and $P\text{-}PO_4$ concentrations in the sediments correlated positively amongst each other (Spearman, $p < 0.0001$, $n = 140$; Table 4). $N\text{-}NH_4$ correlated positively with phosphorous fractions of the sediment (Spearman, $p < 0.02$, $n = 140$). $N\text{-}NO_3$ concentrations in the sediment increased with mean grain size, while phosphorous fractions decreased (Table 4).

At all stream types except FN, the sediment surface featured significantly higher $N\text{-}NO_3$ concentrations than the deeper sediment layers (Wilcoxon test, FR: $p = 0.003$; DI: $p = 0.0003$; DB: $p = 0.0002$; $n = 95$) with a maximum of 37µg g^{-1} dry sediment at DB. Regarding phosphorous fractions, DB showed significantly higher concentrations in the surface layer at all parameters measured (Wilcoxon test, $p < 0.05$; $n = 95$). The upper sediment layer of FN was significantly higher in $P\text{-}PO_4$ concentrations than the deeper sediment layer (Wilcoxon test, $p = 0.01$; $n = 95$). Significant differences in mean grain sizes between sediment layers were found at FN (Wilcoxon test, $p = 0.02$; $n = 95$) with larger fractions prevailing in the upper layer. No differences in OM content of the sediment layers could be determined.

Table 3. Mean values of sediment parameters of the sampling sites in the Weinviertel (Austria) with standard deviation in parentheses. (n = 120, stable istotope analyses n = 60)

† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.

‡ FN, forested reaches with a natural riparian buffer; FR, forested reaches with a restored riparian buffer; DB, degraded broadened reaches without riparian forest buffers; DI, degraded incised reaches without riparian forest buffers; IP, inorganic phosphorous; n.d., not determined; OM, organic matter; OP, organic phosphorous; TP, total phosphorous.

sites †	reach type ‡	depth of oxic layer	mean grain size	OM	TP	OP	IP	P-PO_4	N-NO_3	N-NO_2	N-NH_4	C:N	N:P	$\delta^{13}C$
		10^{-2} m	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment			‰
Hip	FN	10	86.1	5.0	718	332	385	2.27	3.84	1.37	203.5	7.68	2.31	-27.09
		(3)	(75.9)	(2.9)	(168)	(168)	(108)	(0.70)	(3.37)	(0.49)	(188.8)	(1.65)	(0.87)	(0.45)
Str I	FR	5	66.2	4.8	981	350	630	2.38	1.53	0.70	162.8	7.88	2.43	-27.12
		(3)	(28.0)	(1.1)	(149)	(112)	(99)	(1.10)	(1.69)	(0.38)	(205.6)	(1.89)	(1.30)	(1.08)
Str II	DB	5	38.2	8.4	1630	690	941	5.12	5.08	1.23	209.4	10.08	2.53	-27.68
		(2)	(2.3)	(1.3)	(505)	(391)	(284)	(2.82)	(4.96)	(0.63)	(282.8)	(1.40)	(0.74)	(0.30)
Stu I	FR	5	36.2	5.5	1079	488	590	3.53	1.75	1.30	191.2	9.74	2.47	-27.24
		(3)	(2.3)	(0.9)	(131)	(136)	(88)	(0.99)	(2.28)	(0.49)	(226.3)	(1.76)	(0.56)	(0.65)
Stu II	DI	4	46.7	4.7	721	320	401	1.79	1.07	1.61	124.1	9.75	3.03	-27.34
		(1)	(3.5)	(2.5)	(158)	(75)	(117)	(0.64)	(1.42)	(1.12)	(147.6)	(2.12)	(1.56)	(0.86)
Hbb I	FN	10	41.6	12.2	1004	613	391	1.23	1.87	1.62	56.4	n.d.	n.d.	n.d.
		(6)	(2.1)	(2.3)	(523)	(364)	(200)	(0.70)	(1.58)	(1.01)	(27.0)			
Hbb II	DI	3	36.8	8.6	1188	645	542	1.50	1.08	0.73	23.0	n.d.	n.d.	n.d.
		(2)	(1.6)	(2.8)	(431)	(362)	(168)	(1.62)	(1.46)	(0.49)	(21.5)			
Hbg I	DB	6	33.2	5.6	1168	403	765	2.48	0.84	1.41	74.3	n.d.	n.d.	n.d.
		(3)	(0.2)	(0.4)	(159)	(151)	(143)	(1.27)	(0.92)	(0.57)	(57.1)			
Hbg II	DI	8	38.8	4.8	998	387	610	3.19	1.03	1.45	17.4	n.d.	n.d.	n.d.
		(3)	(4.9)	(1.1)	(237)	(150)	(247)	(1.93)	(1.26)	(0.58)	(4.1)			

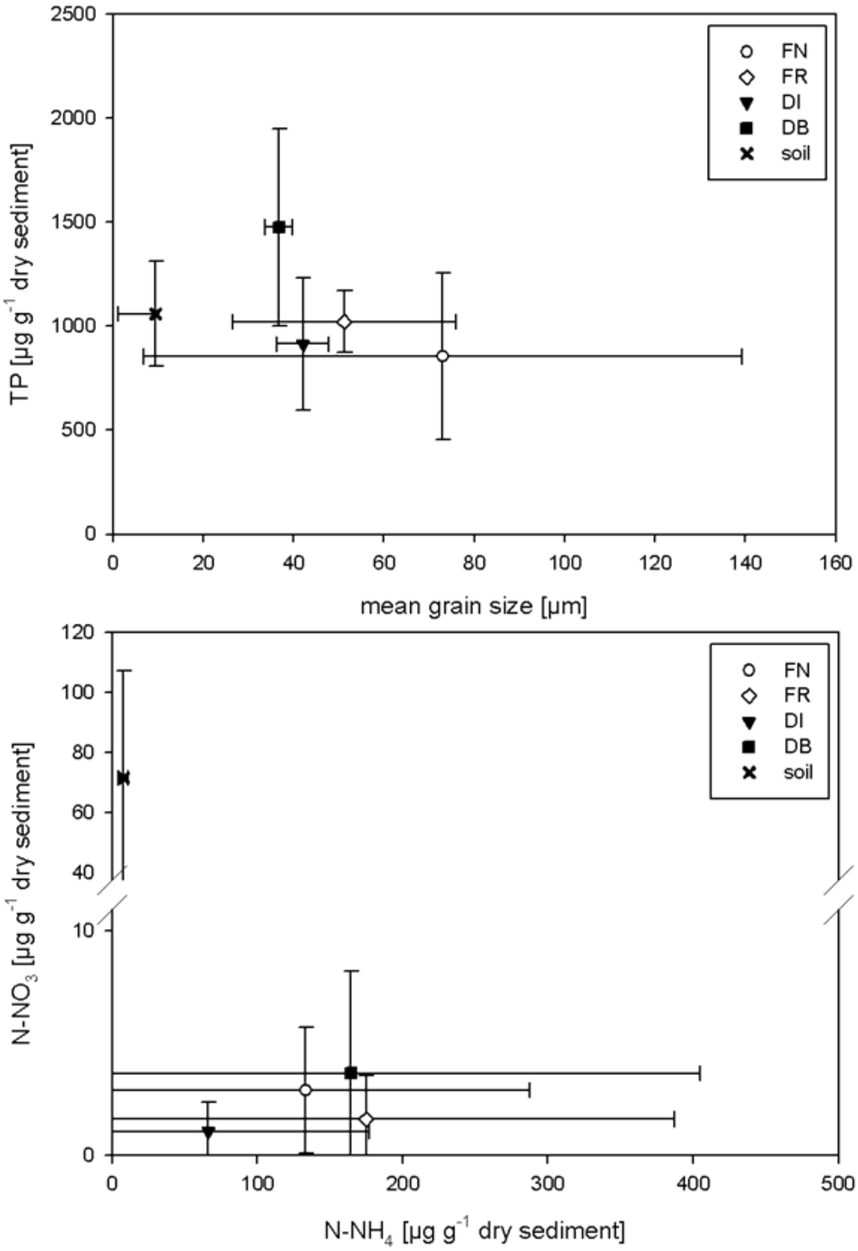

Figure 3. TP, N-NO_3, and N-NH_4 concentrations (given as µg g^{-1} dry sediment) as well as mean grain size [µm] of the different morphological stream types of agricultural streams from the Weinviertel (Lower Austria). For comparison, nutrient concentrations of soil samples from the surrounding areas are shown (Gaitzenauer, 2012).
The upper panel shows mean grain sizes and TP concentrations of soil and sediment samples featuring similar TP concentrations although grain sizes differ. The lower panel displays N-NO_3 and N-NH_4 concentrations of soil and sediment samples. High N-NH_4 concentrations and low N-NO_3 concentrations are found within sediments, whereas soil samples from the surrounding feature the reverse pattern.
FN, natural forested reach; FR, restored forested reach, DB, broadened degraded reach; DI, incised degraded reach; TP, total phosphorous.

INTERACTIONS AMONG HYDROMORPHOLOGY, SEDIMENT AND WATER

In general the upper sediment layer featured more significant correlations to hydromorphological parameters than the pooled sediment samples. The mean grain sizes of the pooled samples correlated positively with stream width, but not with any other hydromorphological parameter (Spearman, $p = 0.04$; $n = 63$; Table 4). The mean grain sizes of the upper sediment layer correlated positively with water depth (Spearman, $p = 0.03$; $n = 54$) and negatively with velocity and the relative coverage of stagnant areas (Spearman, $p = 0.04$ and 0.001, respectively; $n = 22$ and 47, respectively; Table 5). Positive correlations of the organic matter content of the upper sediment layers were observed with bankfull width, the relative coverage of stagnant areas and the CV of velocity (Spearman, $p = 0.03$, 0.02, and 0.04, respectively; $n = 40$, 23, and 48, respectively; Table 5). The organic matter content of the pooled samples correlated positively only with the relative coverage of stagnant areas (Spearman, $p = 0.001$; $n = 54$; Table 4).

The mean grain size correlated negatively with water $P\text{-}PO_4$ and DOC concentrations regarding integrated sediment samples (Spearman, $p < 0.0001$ and $p = 0.003$, respectively; $n = 75$ and 43, respectively; Table 4) as well as upper sediment layers (Spearman, < 0.0001 and $p = 0.02$, respectively; $n = 55$ and 31, respectively; Table 5). DOC was positively correlated to TP and OM content of the upper sediment layers (Spearman, $p < 0.0001$ and $p = 0.03$, respectively; $n = 31$, respectively; Table 5). The $P\text{-}PO_4$ concentrations of the water correlated positively with TP and $P\text{-}PO_4$ concentrations of the sediment concerning pooled sediment samples (Spearman, $p = 0.001$, respectively; $n = 132$; Table 4) and upper sediment layers (Spearman, $p < 0.0001$ and $p = 0.003$, respectively; $n = 84$, respectively; Table 5). $N\text{-}NH_4$ concentration of the water was positively correlated to TP concentrations and OM content of integrated sediment samples (Spearman, $p = 0.01$ and 0.02, respectively; $n = 132$ and 75, respectively; Table 4) as well as TP concentrations and OM content of the upper sediment layers (Spearman, $p < 0.0001$ and $p = 0.007$, respectively; $n = 84$ and 56, respectively; Table 5). Water $N\text{-}NO_3$ concentrations also correlated positively with sediment $P\text{-}PO_4$ concentrations (Spearman, whole sediment sample: $p = 0.004$; $n = 132$; Table 4; upper sediment layers: $p = 0.04$; $n = 84$; Table 5).

Table 4. Correlations among hydromorphological parameters, sediment parameters (pooled sediment samples) and water chemistry given as Spearman's rank correlation coefficients and probability levels.
* $p< 0.05$.
** $p < 0.01$.
† OM, organic matter; CV, coefficient of variation; DOC, dissolved organic carbon; n.s., not significant.

	mean grain size	OM†	sediment $N\text{-}NO_3$	sediment $N\text{-}NH_4$	sediment $P\text{-}PO_4$	sediment TP	water $N\text{-}NO_3$	water $N\text{-}NH_4$	water $P\text{-}PO_4$	DOC
water depth	n.s.	n.s.	n.s.	**-0.19***	n.s.	**0.23****	n.s.	n.s.	n.s.	n.s.
stream width	**0.29***	n.s.	**0.34****	n.s.	n.s.	**0.22***	**0.19***	n.s.	n.s.	**-0.55****
bankfull width	n.s.	n.s.	**0.25***	n.s.	**0.23***	**0.23***	n.s.	n.s.	n.s.	**-0.68****
velocity	n.s.	n.s.	**-0.23***	n.s.	n.s.	n.s.	n.s.	**-0.23***	n.s.	n.s.
flow obstructions	n.s.	n.s.	**0.36****	n.s.	n.s.	**-0.39****	n.s.	n.s.	n.s.	n.s.
stagnant areas	n.s.	**0.43****	**0.19***	**0.35****	n.s.	**0.29****	n.s.	n.s.	n.s.	n.s.
oxic layer	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	**0.31***	n.s.
CV depth	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	**-0.24****	n.s.	n.s.	n.s.
CV width	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	**-0.30***	n.s.	n.s.
CV velocity	n.s.	n.s.	n.s.	n.s.	n.s.	**0.26***	n.s.	n.s.	n.s.	n.s.
mean grain size		**-0.37****	**0.42****	n.s.	**-0.27***	**-0.47****	**0.27***	n.s.	**-0.42****	**-0.44****
OM†	**-0.37****		n.s.	n.s.	n.s.	n.s.	**-0.26***	**0.28***	n.s.	n.s.
sediment $N\text{-}NO_3$	**0.42**.**	n.s.		n.s.	**0.34****	**0.28****	**0.29****	n.s.	n.s.	**-0.29***
sediment $N\text{-}NH_4$	n.s.	n.s.	n.s.		**0.26****	**0.19***	n.s.	n.s.	**0.18***	n.s.
sediment $P\text{-}PO_4$	**-0.27***	n.s.	**0.34****	**0.26****		**0.55****	**0.25****	n.s.	**0.30****	n.s.
sediment TP	**-0.47**.**	n.s.	**0.28****	**0.19***	**0.55****		**0.24****	**0.22****	**0.29****	n.s.

Table 5. Correlations among hydromorphological, water chemistry, and chemical parameters of the upper sediment layers given as Spearman's rank correlation coefficients and probability levels.
* p< 0.05.
** p < 0.01.
† OM, organic matter; CV, coefficient of variation; DOC, dissolved organic carbon; n.s., not significant.

	mean grain size	OM†	sediment $N\text{-}NO_3$	sediment $N\text{-}NH_4$	sediment $P\text{-}PO_4$	sediment TP
water depth	**0.31***	n.s.	**0.29***	n.s.	**0.25***	n.s.
stream width	n.s.	n.s.	**0.39****	n.s.	**0.41****	**0.27***
bankfull width	n.s.	**0.35***	**0.44****	n.s.	**0.57****	**0.39****
flow velocity	**-0.43***	n.s.	n.s.	n.s.	n.s.	n.s.
stagnant areas	**-0.43****	**0.35***	**0.29****	**0.29***	**0.58****	**0.40****
flow obstructions	n.s.	n.s.	n.s.	n.s.	n.s.	**-0.37***
oxic layer	n.s.	n.s.	n.s.	n.s.	**0.37***	n.s.
CV depth	n.s.	n.s.	n.s.	n.s.	**-0.28****	**-0.27***
CV width	n.s.	n.s.	**-0.51****	n.s.	**-0.46***	**-0.41***
CV velocity	n.s.	**0.43***	n.s.	n.s.	n.s.	n.s.
DOC	**-0.35***	**0.39***	n.s.	n.s.	n.s.	**0.66****
water $N\text{-}NO_3$	n.s.	n.s.	n.s.	n.s.	**0.23***	n.s.
water $N\text{-}NH_4$	n.s.	**0.36**.**	n.s.	n.s.	n.s.	**0.40****
water $P\text{-}PO_4$	**-0.57**.**	**0.57****	n.s.	n.s.	**0.32****	**0.43****

Concerning the impact of FRB reaches (FN and FR) on the sediments of degraded downstream reaches, no effects could be detected in any of the parameters measured. Degraded reaches (DI and DB) showed patterns characteristic of the morphological stream type, regardless of the situation in upstream reaches.

We observed significant temporal differences in sediment nutrient concentrations at all sites (Figure 4). The $N\text{-}NO_3$ concentrations in the sediments were highest in spring, whereas $N\text{-}NH_4$ concentrations peaked in summer (Kruskal-Wallis test, $p < 0.05$). All phosphorous fractions showed slightly higher concentrations during summer (mean values of 1.76 mg TP g^{-1} dry sediment and 4.8 µg $P\text{-}PO_4$ g^{-1} dry sediment), but significant seasonal differences were only detected for inorganic phosphorous concentrations (Kruskal-Wallis test, $p < 0.05$, $n = 140$). The variation of phosphorous concentrations was highest in summer. The mean OM content of the sediments featured lowest levels during summer (Figure 4).

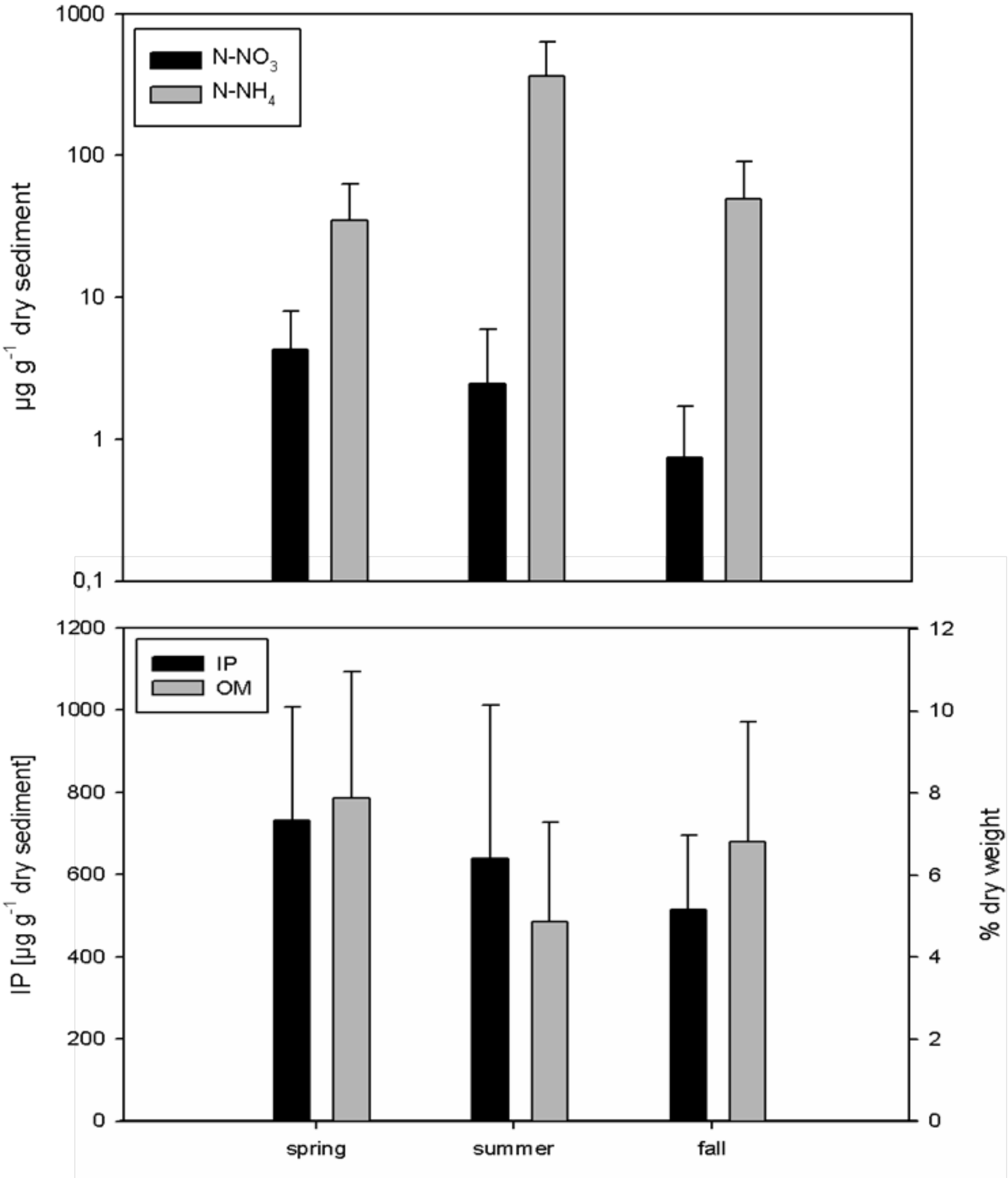

Figure 4. Mean N-NO_3, N-NH_4, and IP concentrations (given as µg g^{-1} dry sediment; n = 140) and standard deviation as well as OM content (given as % dry weight; n = 80) of the sediment per season, measured in agricultural streams in the Weinviertel (Lower Austria). All samples of all sample streams were pooled. IP, inorganic phosphorous; OM, organic matter.

BENTHIC MICROBIAL RESPIRATION

In general, benthic microbial respiration rates increased with increasing temperatures (Figure 4). Nevertheless, significant differences between temperature treatments were only detected at degraded reaches (DI and DB), but not at FRB reaches. At DI, benthic microbial respiration rates measured at 26 °C differed significantly from rates measured at 10 and 18 °C (Mann-Whitney U-test, $p = 0.01$ and 0.07, respectively; $n = 20$). At DB, the 10 °C treatment differed significantly from 18 and 26 °C treatment (Mann-Whitney U-test, $p = 0.03$ and 0.004, respectively; $n = 20$).

Among samples incubated at 10 °C, benthic microbial respiration rates of DI differed significantly from FN and DB rates (Mann-Whitney U-test, $p < 0.0001$, respectively; $n = 20$). Rates measured after an 18 °C treatment were highest at DB and differed significantly from all other stream types (Mann-Whitney U-test, DB-FN: $p = 0.002$, DB-FR: $p = 0.03$, DB-DI: $p < 0.0001$; $n = 20$). Benthic microbial respiration rates measured at DI also differed significantly from rates measured at FN (Mann-Whitney U-test, $p = 0.02$; $n = 20$). At the 26 °C treatment the DB samples differed significantly from all other samples (Mann-Whitney U-test, DB-FN: $p = 0.04$, DB-FR: $p = 0.02$, DB-DI: $p = 0.04$; $n = 20$).

Highest and lowest benthic microbial respiration rates were observed at degraded reaches (DI and DB; mean values of 10.5 and 18.9 g O_2 m^{-2} d^{-1}, respectively). Highly significant positive correlations were detected between IP concentrations and OM content of the sediments and benthic microbial respiration rates (Spearman, $p < 0.0001$ and $p = 0.007$, respectively and $n = 95$ and 60, respectively). Further positive correlations with respiration rates were shown by sediment $N\text{-}NH_4$ concentrations (Spearman, $p = 0.02$; $n = 95$). Water $P\text{-}PO_4$, $N\text{-}NH_4$, and DOC concentrations featured highly significant correlations with benthic microbial respiration rates (Spearman, $p < 0.0001$, $n = 148$, 148, and 124, respectively).

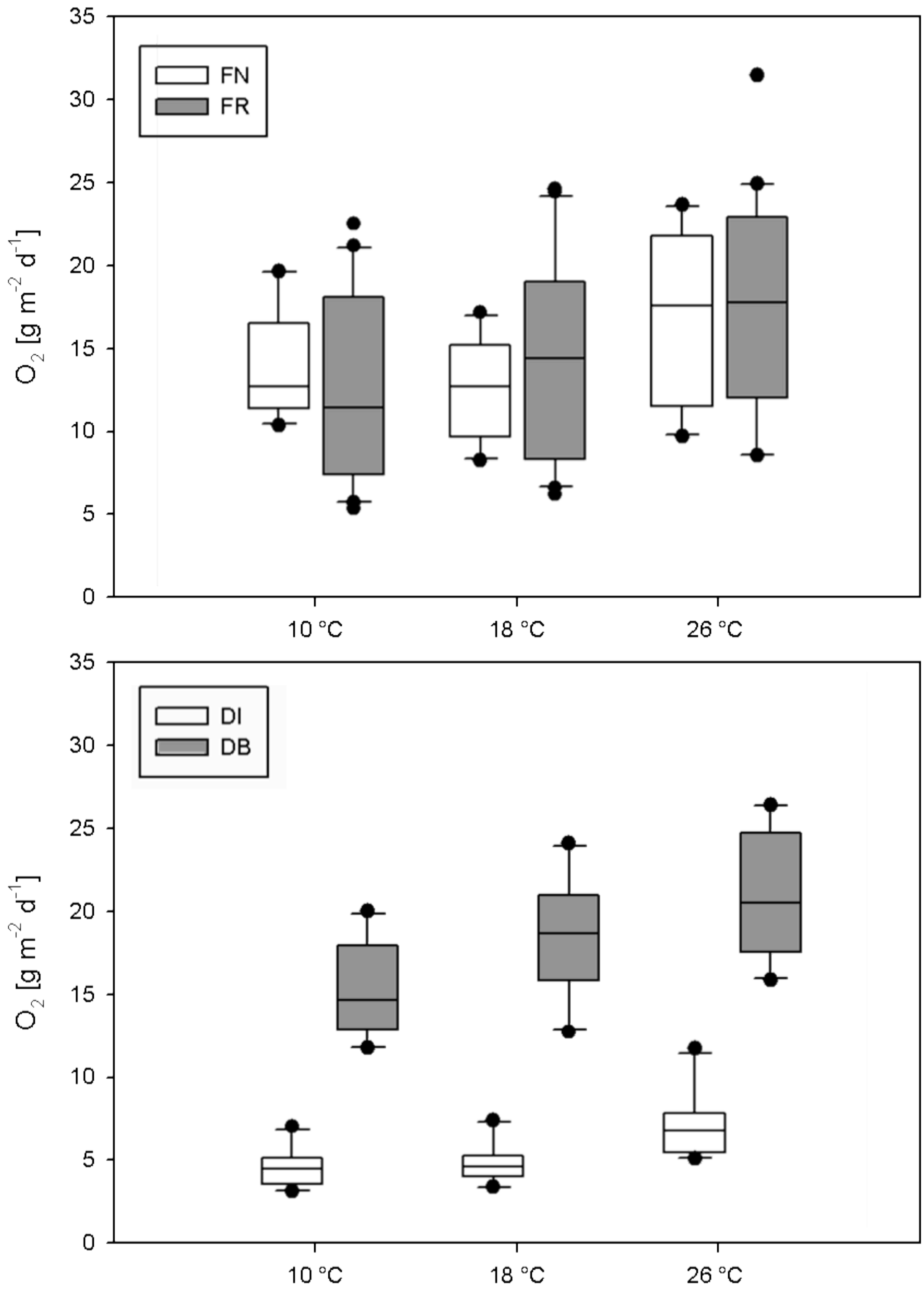

Figure 5. Benthic microbial respiration rates at FRB reaches (FN and FR) and degraded reaches (DI and DB), given as g O_2 m^2 d^{-1} consumption. The vertical axis displays 3 different temperature treatments. n = 45 per site and temperature

FN, natural forested reach; FR, restored forested reach; FRB, forested riparian buffer; DB, broadened degraded reach; DI, incised degraded reach.

DISCUSSION

HYDROMORPHOLOGY

Due to apparent differences in the morphology of the streams, we expected to find significant differences in hydromorphological parameters between different stream types, especially with regard to measures of channel heterogeneity. However, our expectations were only partly confirmed. Stream types differed significantly in mean water depth, bankfull width, and current velocity. Degraded reaches covered both ends of the spectrum, with degraded broadened reaches (DB) featuring the widest and deepest channels, and degraded incised reaches (DI) showing the smallest, shallowest, and fastest flowing channels. FRB reaches were similar in hydromorphology regardless of their origin. However, channel heterogeneity, expressed as of variation of the hydromorphological parameters (depth, width, and velocity), was high in both FRB and degraded reaches. Meandering channels showed a similar cross-sectional variability as straightened channels. This can be explained by the small size of the studied streams, because even the input of small soil agglomerates can raise the heterogeneity of the channel bed (Weigelhofer et al., 2011a). Small-scale irregularities caused by bank erosion, submersed grass vegetation or flooded reed trunks may increase the heterogeneity even of straightened streams when stream width is less than 1 m. Similar findings were described by Weigelhofer et al. (2008) after the investigation of 16 streams of the Weinviertel. However, when comparing longitudinal heterogeneity, forested and degraded reaches differed significantly in the number of woody debris flow obstructions and the percent coverage of stagnant areas.

Water chemistry parameters featured stream specific patterns and were not influenced by hydromorphological conditions of the sampling sites, like stream width, depth or flow velocity. However, Shields et al. (2010) described higher TN and TP concentrations at incised channels, which is opposite to our findings (Table 3, Figure 3). Water nutrient concentrations were high compared to natural, pristine streams. We measured maximum $P\text{-}PO_4$ concentrations of up to 1 mg l^{-1} and $N\text{-}NH_4$ concentrations of more than 1.3 mg l^{-1} in the Weinviertel. Maximum $N\text{-}NO_3$ and DOC concentrations amounted to 36 mg l^{-1} and 28 ppm, respectively. The Weinviertel streams featured similar nutrient concentrations as agricultural streams at the Great Lakes Region, which were tenfold higher than those of pristine streams (Bernot et al., 2006).

SEDIMENT COMPOSITION

Grain sizes were small in general (50 µm), which was expected due to intensive agricultural land use in the catchment and the resulting high erosion potential, leading to a decrease in mean grain size. These findings were supported by C:N and $\delta^{13}C$ values (9 and -27, respectively), which are typical for organic matter of terrestrial origin (Brady, 1990). The variability of the ratios measured was low, which points to the fact that sediment organic matter is more or less of the same origin. Analyses of soil samples (Gaitzenauer, 2012) from adjacent terrestrial sites of the study streams resulted in mean grain sizes of 9 µm. Thus, it can be concluded that the stream sediments contain a high amount of eroded soil originating from the surrounding fields. These findings are also confirmed by observation of massive soil erosion in the adjacent surrounding of the streams.

Differences in mean grain sizes were small among reaches. However, meandering reaches with riparian buffers (FRB) featured larger grain sizes than degraded reaches, as expected, and also showed a higher variability in grain sizes, exhibiting even sand (up to 50 %) to small gravel (up to 19 %). Concerning sedimentary nutrients, the results did not fully meet our expectations. We found generally higher nutrient concentrations at the degraded broadened reach (DB) than at the forested reaches, but differences were significant only for phosphorous fractions, although other authors (Osborne and Kovacic, 1993; Lowrance et al., 1997) have mentioned, that FRBs are more effective in N retention that in P retention (Embarras river and rivers of the Chesapeake Bay watershed, respectively).

We assume that the high overall nutrient concentrations at DB, are due to increased sedimentation within the broadened and reed-vegetated channel. Especially inorganic phosphorous is available at higher concentrations as grain size decreases. This is due to its adsorption properties to the particle surface and as the grain size is lowest at DBs, the high concentrations in P are not surprising. On the other hand, we observed relatively low nutrient concentrations at the degraded incised channels. Grain sizes were also larger there than at DBs. This shows that degraded reaches are highly variable and that sediment structure and composition is mainly a function of particle transport and retention within the respective channel section, which coincides with findings of organic matter retention in other stream systems. (e.g. Young et al., 1978; Speaker et al., 1984). Semi-natural hydromorphology with meandering reaches and riparian buffers showed minor improvements of the sedimentary nutrient status, of the agricultural streams, because nutrient concentrations were slightly lower. Furthermore, we did not find any effects of FRB reaches on the sediment quality of downstream reaches, neither in grain size nor in nutrient concentrations.

In contrast to the findings of Osborne and Kovacic (1993), we found seasonal variations in $N\text{-}NO_3$ concentrations in the sediment. Highest $N\text{-}NO_3$ concentrations were observed in spring samples, especially within the upper sediment layers, indicating fertilizer application (Lefebvre et al., 2007). According to Montreuil et al. (2010), nitrogen surplus of agricultural origin constitutes the main factor controlling nitrate concentrations in agricultural streams. In general, $N\text{-}NO_3$ concentrations of sediments were low, ranging from 0.01 to 14.9 µg g^{-1} dry sediment, whereas $N\text{-}NH_4$ concentrations were high, especially in summer, amounting up to 1 mg g^{-1} dry sediment. Interestingly, $N\text{-}NO_3$ and $N\text{-}NH_4$ concentrations of the water samples featured a reversed pattern, with high nitrate and low ammonium concentrations (5.5 mg l^{-1} and 104 µg l^{-1}, respectively). Nutrient concentrations of the sediment samples were also compared with soil samples from the surrounding crop fields (Gaitzenauer, 2012). We noticed equal phosphorous concentrations in the soil and the stream sediments, but nitrogen concentrations featured clear differences. $N\text{-}NO_3$ concentrations were tenfold higher in the soil than in the sediments, whereas $N\text{-}NH_4$ concentrations were hundredfold smaller. This patterns could be explained by anoxic processes in the organic-rich sediments, like denitrification or dissimilatory reduction of nitrate to ammonium (Revsbech, et al. 2006), resulting in $N\text{-}NO_3$ depletion and $N\text{-}NH_4$ accumulation in the sediments. Furthermore, $N\text{-}NH_4$ accumulates due to mineralization of organic N under hypoxic conditions (Lefebvre et al., 2004), which applies for degraded streams in the Weinviertel.

While soil samples from the surrounding crop fields showed no seasonal differences in phosphorous content, bank soil samples peaked in autumn, probably due to leaf fall (Gaitzenauer, 2012). Thus, we also expected high concentrations of organic matter and concurrent high organic phosphorous amounts in the sediments, especially as fertilizers applied in spring do not usually contain phosphorous in the investigated catchment. Nevertheless, phosphorous concentrations in the sediments were highest in spring. Especially inorganic phosphorous displayed a distinct seasonal variability with lowest values in the autumn samples. A possible explanation for this pattern could be the retention of organic particles on the banks which might result in a time lag between concentrations in the topsoil layer of the banks and those in the stream sediments. The seasonal patterns found for phosphorous are in concordance with other authors (Osborne and Kovacic, 1993) who also mentioned higher TP concentrations in spring due to leaching and decomposition of organic matter. In general, the differences in OM content between morphological stream types were quite low despite the different bank vegetation. This could be due to the fact that agricultural streams are exposed to many external pressures in the catchment, like agricultural land-use activities or bank management measures (e.g. reed cutting). Such interventions could overrule natural, seasonal in-stream processes (Lefebvre et al., 2007).

BENTHIC MICROBIAL RESPIRATION

In contrast to our expectations, respiration rates were high at all morphological stream types yielding values between 10 and 20 g O_2 m^{-2} d^{-1}. Only the sediments of the degraded incised channel showed significantly smaller respiration rates (5-7 g O_2 m^{-2} d^{-1}). Bunn et al. (1999) stated that high benthic respiration rates may result from small grain sizes and high nutrient background. At the Weinviertel, even the lowest benthic respiration rates were about tenfold higher than those measured by Fellows et al. (2006) in other fine sediment substrates. In addition, Parkhill and Gulliver (2002) noticed that whole-stream respiration decreased with increasing fine sediment loading. On the other hand, Gücker and Pusch (2006) measured community respiration in eutrophic lowland streams and observed even higher rates than in the Weinviertel (10 to 60 g O_2 m^{-2} d^{-1}). This shows that high benthic respiration rates seem to be characteristic for agricultural streams with a high trophic background and accumulations of organic matter in the channel. In our study, respiration rates increased with organic matter and nutrient content in the sediment as well as in the overlying water column.

Benthic microbial respiration increased with increasing temperature, which is the main environmental driver for benthic microbial respiration (Doering et al., 2011). While in the Weinviertel, the degraded broadened reach featured the highest increase in benthic microbial respiration with increasing temperature, the forested natural reach showed the weakest reaction to the temperature increase. This may be due to an adaption of the respective benthic microbial community which is exposed to rather high summer temperatures of > 30°C at the open, broadened reach, while the forested reach revealed a more buffered temperature regime with summer maxima < 30°C.

IMPLICATIONS FOR MANAGEMENT OF AGRICULTURAL LOW-ORDER STREAMS

It is known, that agricultural streams are heavily impacted by soil and concurrent nutrient loading through erosion in the catchment (Walling, 1999; Bernot et al., 2006). The plantation of FRBs is a well-known method to mitigate this pressure (Lowrance et al., 1984; Craig et al., 2008). Such measures are also frequently applied to Austrian low-order streams within agricultural catchments to improve structural and ecological conditions (Weigelhofer et al., 2011a). However, this study shows that sometimes these measures are not able to improve sediment quality sufficiently.

The reason is that agricultural low-order streams are exposed to soil erosion, which load the streams with high amounts of fine particles. Therefore the soil accumulates on the channel bed and these depositions influence channel heterogeneity as well as sediment structure and alters the oxygen supply of the channel bed by clogging the interstices (Wood and Armitage, 1999). The appearance of enduring anoxic conditions in the sediment influences natural nutrient cycling (Boulton, 2007). Our results imply that low-order streams, especially in dry agricultural catchments are not able to cope with the massive load of eroded material. Deposited material could be transported during times of high current velocities, which are usually only available during floods. On the other hand, that is when erosion peaks, thus the deposited material might just be replaced. In Austria, the restoration of low-order streams often implies the reconstruction of meandering stream courses and the plantation of riparian buffer strips. Due to these FRBs, channel heterogeneity and flow patterns could be increased by woody debris dams and meanders. Thus, the retention of fine particles might be further increased resulting in higher deposition of eroded material within the channel bed. Furthermore we found out that FRBs do not have any effect on sediment quality of degraded downstream reaches, which implies that gaps in FRBs compensate possible positive effects of FRBs, because such gaps facilitate direct pathways for soil and nutrient inputs.

Many authors (e.g. Montreuil et al., 2010) have described the ability of FRBs to reduce lateral loading of soil and concurrent nutrients. Nevertheless, our study indicates that FRBs still allowing direct soil inputs could not significantly contribute to improvements in sediment quality of agricultural low-discharge streams. Therefore, the preferable way to improve sediment quality in agricultural low-order streams, is to minimize soil erosion at the catchment scale and consider the reduction of direct soil inputs in the buffer zone design (Wood and Armitage, 1999; Gücker and Pusch, 2006; Craig et al., 2008).

To ensure sufficient erosion precautions, initially, hot spots and flow paths of eroded particles have to be identified. In a next step adequate retention structures like basins or riparian buffer strips have to be established. Agricultural best management practices (BMPs) have to be applied in the adjacent surrounding to reduce soil input and to raise the effectiveness of FRBs (Baillie, 1995; Craig et al., 2008). Furthermore, these measures as well as their effect on stream ecosystems have to be monitored (Bernhardt et al., 2007). Nevertheless, there are still many questions that are unanswered, because watershed process dynamics are still not completely understood (Wohl et al., 2005) and thus long-term studies are essential to allow for efficient restoration measures of streams in agricultural catchments.

CONCLUSION

Forested riparian buffer zones are usually more effective at high order streams due the larger buffer zone areas (Montreuil et al., 2010). On the other hand, small streams exhibit a high potential to improve water quality (Lowrance et al., 1997; Alexander et al., 2000), especially during low and moderate flow (Craig et al., 2008), and are closely interlinked with the terrestrial catchment. Therefore, the creation of riparian buffers along headwaters is essential, especially within agricultural catchments, because they immediately mitigate the inputs efficiently (Naiman and Décamps, 1997). Forested riparian buffer zones usually maintain high amounts of wood and CPOM input, as confirmed by the increased number of woody flow obstructions and the proportion of stagnant areas found on the channel bed at our FRB reaches. Flow obstructions decrease water velocity and increase hydraulic retention (Roberts et al., 2007), which may result in the enhancement of natural processes like nutrient retention (Aldridge et al., 2009)

Our study streams featured similar slopes (0 to 2 %) and ranges of discharge, with 0.1 to 5 l s^{-1} at low discharge and 10 to 20 l s^{-1} at medium discharge. Under these conditions, current velocities are low and there is hardly any mobilization of sediments. Due to the low infiltration capacity of the soil and the drainage of the catchment, flashy floods are frequent in the headwaters of the Weinviertel. During flood events, the deposited sediment is flushed out, especially at straightened, incised channel sections without flow obstructions. On the other hand, floods load streams with new sediments mainly derived from soil erosion in the adjacent agricultural areas. As our study shows, different morphological stream types react differently to this repeated erosion and sedimentation events. At straightened, incised channel sections, the whole fine sediment layer is mobilized during floods. At broadened, reed-vegetated channels only the sediment surface is affected due to the hydraulic resistance of the reed, shown by the small grain sizes and high organic matter accumulations, which indicate a distinct sedimentation of suspended material within the reed. These findings were further supported by the positive correlation between the organic matter content and the extent of stagnant areas in the investigated stream reaches.

At FRB reaches, fine sediment accumulations occur on the stream bed, but are not evenly distributed as in straightened channels because alternating deposition and erosion zones are established along the meandering stream course. An increase in longitudinal channel heterogeneity, by providing physical structures such as debris dams or submerged roots, also facilitates the formation of riffle-pool-sequences (Montgomery et al., 1995). These depositional and erosional zones have a sorting effect on the sediments (Nakamura and Swanson, 1993; Wood and Armitage, 1999), because particle retention is driven by different mechanisms at pools and riffles (Speaker et al., 1984) as was also shown by the higher variability in grain sizes at the FRB reaches.

Our study shows that FRB reaches may result in an improvement in sediment structure and composition even in highly impacted agricultural streams, especially in the diversity of choriotopes. Furthermore, we pointed out that non-continous FRBs along agricultural streams are detrimental to downstream sections, because their effects are spatially restricted. We could also show that due to the elevated nutrient background and the high loadings of eroded soil from the agricultural surroundings, sediment structure and composition in agricultural streams can only be significantly improved by additional measures at catchment scale, which reduce soil erosion (Wood and Armitage, 1999) and by means of best management practices in agriculture (BMPs) (Baillie, 1995; Craig et al., 2008; Gücker and Pusch, 2006).

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ANNEX

Annex 1. Hydromorphological data of the sampling sites in the Weinviertel (Lower Austria).
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡ CV, coefficient of variation.

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Hip	3	0.90	2.00	0.09	50	90	0.68	0.19	0.65
Hip	5	0.90	0.85	0.07	25	40	0.71	0.27	0.83
Hip	2	0.98	1.40	0.15	20	50	1.17	0.39	1.05
Hip	5	0.77	1.30	0.07	25	32	0.86	0.31	0.78
Hip	9	0.62	1.30	0.05	10	50	0.80	0.24	0.65
Hip	3	1.10	1.25	0.11	50	45	0.73	0.07	0.64
Hip	22	1.37	2.61		35	80	0.63		
Hip	34	0.61	1.03		20	30	0.54		
Hip	7	1.16	1.70		10	32	0.64		
Hip	16	1.08	1.30		15	50	0.75		
Hip	9	1.32	1.40			45	0.51		
Hip	13	0.94	1.38			40	0.53		
Hip	12	1.45	1.65			45	0.95		
Hip	30	2.00				60	1.19		
Hip	17	0.75				35	0.58		
Hip	9	1.16					0.78		
Hip	5	1.05					0.78		
Hip	13	0.53					0.67		
Hip	6	0.87					0.76		
Stu I	6	0.92	1.00	0.07	15	55	0.58	0.05	1.29
Stu I	6	0.85	1.30	0.02	15	45	0.75	0.29	1.47
Stu I	8	0.89	1.26	0.05	15	30	0.68	0.20	1.30
Stu I	4	0.81	1.27	0.04	10	65	0.70	0.16	0.91
Stu I	8	0.84	1.45	0.03	15	45	0.62	0.18	0.98

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Stu I	6	0.42	1.15	0.11		65	0.68		1.33
Stu I	4	0.80	1.20	0.06		45	0.57		1.39
Stu I	3	0.63	0.98	0.10		50	0.66		0.68
Stu I	4	0.60	1.00	0.09		65	0.55		0.94
Stu I	4	0.68	1.00	0.15		45	0.61		0.98
Stu I	5	1.09	1.17	0.19		35	0.84		0.86
Stu I	9	1.04	1.10	0.07		52	0.67		0.99
Stu I	8	0.80	1.14	0.24		58	0.68		0.94
Stu I	6	0.83				55	0.86		
Stu I	8	0.66				35	0.74		
Stu I	8	0.90					0.69		
Stu I	15	1.10					0.67		
Stu I	8	0.80					0.92		
Stu II	10	0.65	0.70	0.14	0	46	0.57	0.23	0.77
Stu II	6	0.43	0.59	0.07	0	35	0.76	0.21	1.45
Stu II	5	0.78	0.79	0.07	3	23	0.57	0.22	1.40
Stu II	5	0.64	0.74	0.08	0	45	0.75	0.22	1.26
Stu II	7	0.92	0.92	0.11	0	20	0.74		0.93
Stu II	7	0.81	0.81		5	40	1.22		
Stu II	6	1.08	1.08		0	55	0.78		
Stu II	9	0.78	0.78		0	50	0.69		
Stu II	7	0.61	0.61			30	0.82		
Stu II						52			
Stu II						25			
Stu II						45			
Stu II						35			
Stu II						50			
Str I	12	0.78	0.92	0.17	5	25	0.70	0.60	1.50
Str I	10	0.90	1.30	0.07	10	40	0.65	0.19	1.20
Str I	16	1.20	1.36	0.11	10	50	0.61	0.05	1.00

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Str I	11	1.10	1.50	0.17	10	32	0.65	0.19	0.96
Str I	14	1.05	1.05	0.09	10	50	0.84	0.06	0.70
Str I	12	1.07	1.22	0.09		45	0.63		0.67
Str I	9	0.94	1.07	0.14		40	0.63		0.70
Str I	10	0.67	0.84	0.14		30	0.72		1.04
Str I	10	0.80	0.98	0.13		32	0.72		0.67
Str I	11	0.82	0.92	0.03		50	0.79		1.12
Str I	7	0.90	1.40	0.02		45	0.95		0.84
Str I	9	0.92	0.95	0.03		40	0.72		0.96
Str I	9	0.90	1.05	0.03		45	0.56		0.85
Str I	14	0.90	0.90	0.03		32	0.62		1.39
Str I	16	1.00	1.05	0.03		35	0.75		0.77
Str I	11	1.16	1.26				0.54		
Str I	6	0.74	0.85				0.97		
Str I	4	0.60	1.25				0.74		
Str I	7	0.62	1.07				0.74		
Str I	5	0.60	1.25				1.03		
Str I	5	0.68	1.20				0.85		
Str I	5	0.64	1.20				0.86		
Str I	4	0.69	1.10				0.88		
Str II	18	1.90	1.95	0.06	0	70	0.59	0.12	1.05
Str II	13	2.05	2.18	0.05	0	60	0.67	0.13	0.96
Str II	15	1.80	2.00	0.07	0	65	0.57	0.18	1.12
Str II	13	1.50	1.65	0.06	0	70	0.63	0.18	0.83
Str II	16	1.65	1.80	0.06	0	50	0.66	0.15	1.11
Str II	12	1.95	2.45	0.08		65	0.78		0.73
Str II	10	1.97	2.31	0.06		50	0.36		0.79
Str II	12	1.70	2.34	0.08		55	0.51		0.80
Str II	11	1.40	2.10	0.08		80	0.37		1.07
Str II	11	1.80	2.40	0.06		70	0.25		0.91

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Str II	85	1.33	2.00	0.04		50	0.56		1.35
Str II	11	1.04	1.40	0.05		60	0.52		0.81
Str II	86	1.30	1.90	0.06		75	0.53		0.76
Str II	8	1.35	1.95	0.05		85	0.51		1.05
Str II	12	0.86	1.50	0.03		85	0.77		1.04
Str II	5	0.52				70	0.95		
Str II	4	0.62				65	0.79		
Str II	3	0.70				75	0.82		
Str II	7	0.65					0.99		
Str II	3	0.85					0.84		
Hbb I	10	0.82	0.90	0.03	35	85	0.66	0.18	0.75
Hbb I	6	0.92	0.84	0.02	25	90	0.60	0.20	0.80
Hbb I	6	0.82	1.10	0.04	15	85	0.59	0.16	0.70
Hbb I	9	0.85	0.90	0.03	20	75	0.67	0.18	0.71
Hbb I	16	1.00	1.00	0.03	25	80	0.57	0.17	0.77
Hbb I	16	1.27	1.32	0.04		80	0.54	0.19	0.76
Hbb I	3	0.40	1.01	0.04		75	0.75		1.14
Hbb I	10	0.90	1.40	0.08		85	0.78		0.92
Hbb I	8	0.68	1.20	0.03		75	0.87		0.80
Hbb I	10	0.70	1.03	0.04		80	0.77		0.74
Hbb I	11	0.95		0.06		75	0.92		1.03
Hbb I	10	1.00		0.05		80	0.76		0.94
Hbb I	6	0.32				85	0.79		
Hbb I	3	0.59				75	0.78		
Hbb I	5	0.62				90	0.74		
Hbb I	4	0.48				81	0.62		
Hbb I	2	0.55							
Hbb I	4	0.40							
Hbb I	3	0.70							
Hbb I	4	0.80							

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Hbb I	2	0.79							
Hbb II	6	0.68	0.80	0.02			0.90	0.14	0.95
Hbb II	9	0.85	0.95	0.04			0.72	0.28	0.90
Hbb II	9	0.87	0.90	0.02			0.76	0.21	1.02
Hbb II	7	0.90	1.13	0.02			0.98	0.19	0.88
Hbb II	6	0.46	1.08	0.04			0.60	0.23	0.74
Hbb II	8	0.30					0.97		
Hbb II	2	0.30					0.76		
Hbb II	2	0.45					0.66		
Hbb II	3	0.35					0.56		
Hbb II	3	0.20					0.63		
Hbb II	2						0.79		
Hbg I	11	1.90	2.94	0.14	0	69	0.66	0.83	0.85
Hbg I	11	2.15	2.82	0.11	0	65	0.59	0.19	1.20
Hbg I	15	1.90	3.90	0.08	0	75	0.71	0.51	0.74
Hbg I	11	1.10	2.20	0.12	0	85	0.69	0.55	0.76
Hbg I	9	0.45	0.50	0.07	0	75	0.68	0.43	0.90
Hbg I	12	0.47	0.58	0.09	0	65	0.77	0.55	0.78
Hbg I	9	0.60	0.80	0.07	0	80	0.67		0.78
Hbg I	6	0.73	0.90			75	0.74		
Hbg I	10	0.60	0.80			55	0.61		
Hbg I	12	0.48	0.70			65	0.76		
Hbg I	7	0.48	0.70			75	0.75		
Hbg I						60			
Hbg I						65			
Hbg I						57			
Hbg I						59			
Hbg I						75			
Hbg I						85			
Hbg I						75			

sites †	water depth	stream width	bankfull width	velocity	woody flow obstructions	stagnant areas	CV depth ‡	CV width ‡	CV velocity ‡
	10^{-2} m	m	m	m s^{-1}	no. per reach	% coverage			
Hbg I						76			
Hbg I						90			
Hbg I						80			
Hbg II	14	0.64	0.78	0.16	2	15	0.63	0.11	0.67
Hbg II	16	0.73	0.70	0.15	2	30	0.65	0.24	0.67
Hbg II	13	0.80	0.74	0.26	2	20	0.63	0.28	0.48
Hbg II	18	0.83	0.80	0.21	0	14	0.63	0.08	0.70
Hbg II	9	0.84	0.87	0.14	0	25	0.89	0.18	0.80
Hbg II	10	0.87	1.10	0.11	0	30	0.59		0.77
Hbg II	4	0.60		0.10		35	0.65		0.86
Hbg II	3	0.40		0.03		5	0.64		1.02
Hbg II	5	0.40		0.11		20	0.48		0.77
Hbg II	4	0.42		0.08		28	0.49		0.99
Hbg II	9	0.56		0.10		25	0.67		0.80
Hbg II	9	0.50		0.04		30	0.70		1.10
Hbg II	13	0.65		0.12		10	0.73		0.81
Hbg II	12	0.40		0.09		30	0.74		0.80
Hbg II	8	0.75		0.12		20	0.76		1.04
Hbg II	6	0.75				12	0.78		
Hbg II	6	0.68				25	0.74		
Hbg II	12	0.70				5	0.76		
Hbg II	4	0.74					0.94		
Hbg II	7	0.60					0.76		
Hbg II	5	0.68					0.88		

Annex 2. Water chemistry data of the sampling sites in the Weinviertel (Lower Austria).
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡ DOC, dissolved organic carbon.

sites †	$P\text{-}PO_4$	$N\text{-}NH_4$	$N\text{-}NO_3$	$N\text{-}NO_2$	DOC ‡
	µg l^{-1}	µg l^{-1}	µg l^{-1}	µg l^{-1}	ppm
Hip	68	64	69456	22	2.78
Hip	69	56	6864	22	2.06
Hip	70	51	6789	23	2.11
Hip	64	33	6768	23	1.58
Hip	64	26	6524	24	1.60
Hip	47	41	6866	11	1.61
Hip	47	39	6810	10	2.23
Hip	46	36	6728	11	2.22
Hip	46	33	6742	11	2.17
Hip	46	35	6840	11	2.36
Hip					2.46
Hip					2.37
Stu I	45	53	7330	80	4.90
Stu I	28	15	3000	0	3.76
Stu I	9	54	4380	40	4.11
Stu I	20	18	33140	10	4.50
Stu I	138	115	4590	140	3.50
Stu I	179	52	4820	40	2.90
Stu I	91	92	4840	120	3.70
Stu I	91	40	4450	100	2.00
Stu I	265	36	3520	20	3.40
Stu I	241	142	5530	70	3.80
Stu I	580	12	720	0	3.60
Stu I	128	1100	210	0	3.50
Stu I	78	60	6900	60	3.39
Stu I	74	47	6660	50	3.80
Stu I	157	178	7990	60	3.85
Stu I	47	17	7700	60	4.06
Stu I	85	60	8060	80	4.00
Stu I	70	11	200	0	3.30
Stu I	53	35	8980	160	2.90
Stu I	28	15	10320	110	3.70
Stu I	125	123	4760	50	1.98
Stu I	85	90	5860	60	3.40
Stu I	103	109	5870	60	3.80
Stu I	85	89	6160	40	3.60
Stu I	98	99	6630	50	3.40
Stu I	46	84	4540	40	3.35
Stu I	24	14	36680	0	3.60
Stu I	81	43	4600	50	3.78
Stu I	116	33	5040	40	6.35
Stu I	73	161	4200	90	3.30
Stu I	56	72	4720	80	3.30
Stu I	226	28	4560	40	2.80

sites †	P-PO_4	N-NH_4	N-NO_3	N-NO_2	DOC ‡
	µg l^{-1}	µg l^{-1}	µg l^{-1}	µg l^{-1}	ppm
Stu I	173	84	5370	50	3.60
Stu I	580	180	660	0	1.90
Stu I	12	1310	160	10	3.30
Stu I	51	42	6390	40	3.70
Stu I	51	36	6340	40	3.50
Stu I	17	29	7360	30	3.30
Stu I	46	12	7390	50	2.91
Stu I	53	24	7670	5	3.50
Stu I	49	30	1752	26	3.08
Stu I	50	31	1752	25	3.09
Stu I	52	33	1742	25	2.92
Stu I	45	41	1720	25	2.89
Stu I	54	27	1712	25	2.92
Stu I	19	42	6378	28	2.98
Stu I	24	36	6155	26	2.96
Stu I	25	35	6329	26	2.97
Stu I	25	35	6267	27	
Stu I	29	57	6440	27	
Stu I	69	68	6283	63	
Stu I	73	89	6528	66	
Stu I	71	74	6399	67	
Stu I	68	76	6305	66	
Stu I	71	63	6435	71	
Stu II	122	18	70	10	4.25
Stu II	46	32	8150	60	8.24
Stu II	41	37	9630	70	3.90
Stu II	11	268	0	10	4.00
Stu II	95	24	5180	50	3.70
Stu II	33	77	5630	50	4.20
Stu II	45	23	5700	50	2.40
Stu II	51	26	5780	50	3.60
Stu II	46	32	6470	50	4.00
Stu II	14	42	5230	60	3.90
Stu II	23	28	16370	10	3.90
Stu II	148	89	7740	70	7.31
Stu II	50	11	2171	21	3.11
Stu II	50	14	2113	22	3.13
Stu II	47	6	1992	23	3.16
Stu II	48	12	1943	25	3.11
Stu II	49	11	1849	29	3.14
Stu II	27	24	5573	24	3.02
Stu II	26	27	5462	23	3.01
Stu II	26	19	5102	22	2.99
Stu II	25	22	5224	21	
Stu II	23	23	5140	22	
Stu II	66	63	6287	65	
Stu II	64	49	5969	61	
Stu II	62	53	5997	58	

sites †	P-PO_4	N-NH_4	N-NO_3	N-NO_2	DOC ‡
	µg l^{-1}	µg l^{-1}	µg l^{-1}	µg l^{-1}	ppm
Stu II	59	52	5837	54	
Stu II	58	64	5858	54	
Str I	42	59	6540	70	4.20
Str I	14	1144	40	0	6.60
Str I	26	55	7180	60	3.60
Str I	24	36	7160	50	5.20
Str I	31	66	9120	70	4.90
Str I	21	34	7970	60	5.10
Str I	29	38	8270	60	4.80
Str I	21	228	70	10	5.03
Str I	68	18	50	0	5.09
Str I	21	18	8610	70	5.75
Str I	17	17	10240	120	5.67
Str I	62	70	5560	60	4.80
Str I	42	50	6580	80	4.83
Str I	51	51	6510	70	4.74
Str I	128	311	13168	66	4.82
Str I	129	284	12657	65	4.95
Str I	131	288	12710	67	4.89
Str I	132	279	12643	67	5.35
Str I	133	271	12985	67	5.21
Str I	110	49	3061	32	5.23
Str I	97	45	3691	31	
Str I	87	114	2596	33	
Str I	83	42	1800	32	
Str I	125	48	4445	31	
Str I	136	92	9186	178	
Str I	111	72	8577	72	
Str I	104	383	5648	258	
Str I	122	645	5746	292	
Str I	113	498	5966	265	
Str I	106	366	5802	244	
Str I	105	291	5665	235	
Str II	60	118	5860	120	4.70
Str II	126	115	6420	150	6.30
Str II	73	193	5980	120	4.60
Str II	26	24	12900	0	5.30
Str II	85	109	5870	130	5.60
Str II	192	88	6030	90	7.30
Str II	202	188	7460	170	5.40
Str II	586	21	760	10	5.56
Str II	111	274	13267	65	5.50
Str II	112	265	14680	65	8.40
Str II	112	263	19346	67	5.10
Str II	113	261	18895	67	6.50
Str II	111	252	18769	66	6.00
Str II	115	252	19125	68	6.98
Str II	100	47	4845	30	6.40

sites †	P-PO_4	N-NH_4	N-NO_3	N-NO_2	DOC ‡
	µg l^{-1}	µg l^{-1}	µg l^{-1}	µg l^{-1}	ppm
Str II	117	24	4162	30	6.11
Str II	109	56	4584	34	5.94
Str II	131	32	3528	33	6.02
Str II	111	46	5182	32	5.49
Str II	78	203	10700	64	5.56
Str II	236	167	4894	150	5.63
Str II	229	184	5310	166	5.68
Str II	238	166	5397	158	5.62
Str II	234	231	5427	162	5.66
Str II	245	163	5344	159	
Hbb I	133	57	8020	80	7.30
Hbb I	408	116	70	10	2.99
Hbb I	73	12	470	10	3.73
Hbb I	119	55	8940	70	3.48
Hbb I	124	22	8230	80	9.19
Hbb I	9	78	775	32	
Hbb I	9	80	860	28	
Hbb I	8	64	1111	17	
Hbb I	11	99	647	35	
Hbb I	12	85	613	41	
Hbb I	432	110	397	43	
Hbb I	423	112	374	42	
Hbb I	399	105	352	45	
Hbb I	393	118	340	42	
Hbb I	393	108	335	40	
Hbb I	7	50	1836	11	
Hbb I	7	51	1901	12	
Hbb I	7	57	1841	13	
Hbb I	7	55	1760	12	
Hbb I	7	58	1690	14	
Hbb II	58	57	6800	40	7.29
Hbb II	61	47	6940	40	6.80
Hbb II	59	49	6910	50	7.63
Hbb II	62	39	6970	40	4.26
Hbb II	77	19	7450	50	3.70
Hbb II	20	17	2400	10	4.00
Hbb II	8	17	6420	40	7.11
Hbb II	17	3	14730	20	6.85
Hbb II	22	64	1288	7	10.00
Hbb II	27	89	1106	12	7.43
Hbb II	29	64	914	16	7.85
Hbb II	28	64	822	14	7.29
Hbb II	29	66	785	15	7.57
Hbb II	387	116	368	40	
Hbb II	410	124	373	41	
Hbb II	410	107	372	41	
Hbb II	396	113	366	40	
Hbb II	392	108	378	40	

sites †	$P\text{-}PO_4$	$N\text{-}NH_4$	$N\text{-}NO_3$	$N\text{-}NO_2$	DOC ‡
	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	ppm
Hbb II	12	50	995	3	
Hbb II	12	32	925	5	
Hbb II	10	25	866	5	
Hbb II	10	27	811	6	
Hbb II	9	28	787	6	
Hbg I	23	7	6540	20	5.28
Hbg I	28	11	2310	0	5.34
Hbg I	23	14	6760	20	7.26
Hbg I	10	37	5870	30	5.10
Hbg I	18	9	11650	0	5.84
Hbg I	91	27	6640	60	6.64
Hbg I	100	40	10050	60	28.00
Hbg I	75	109	8330	40	5.20
Hbg I	20	21	14130	10	
Hbg I	1076	102	2231	141	
Hbg I	1022	111	2107	134	
Hbg I	1054	106	2657	138	
Hbg I	957	113	2667	132	
Hbg I	970	117	2049	122	
Hbg I	568	77	3799	55	
Hbg I	568	31	3641	48	
Hbg I	574	41	3615	48	
Hbg I	564	38	3647	47	
Hbg I	562	34	3632	47	
Hbg I	716	36	3611	46	
Hbg I	489	412	1706	135	
Hbg I	486	415	1765	133	
Hbg I	478	416	1779	133	
Hbg I	474	419	1768	131	
Hbg II	78	105	8780	50	5.76
Hbg II	91	67	12540	50	5.08
Hbg II	88	63	14830	60	7.79
Hbg II	549	11	2870	10	5.60
Hbg II	9	1240	0	10	6.05
Hbg II	87	86	7210	60	7.14
Hbg II	95	76	7280	60	7.40
Hbg II	23	32	8350	30	5.40
Hbg II	974	65	3122	119	
Hbg II	973	67	3020	115	
Hbg II	961	62	3111	116	
Hbg II	957	82	3002	119	
Hbg II	942	71	3061	118	
Hbg II	362	25	2608	46	
Hbg II	488	39	2632	46	
Hbg II	492	56	2672	46	
Hbg II	314	30	2717	46	
Hbg II	402	23	2763	48	
Hbg II	753	89	3004	120	

sites †	$P\text{-}PO_4$	$N\text{-}NH_4$	$N\text{-}NO_3$	$N\text{-}NO_2$	DOC ‡
	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	$\mu g\ l^{-1}$	ppm
Hbg II	784	82	3072	119	
Hbg II	766	81	3096	118	
Hbg II	714	87	3076	111	
Hbg II	774	84	3113	122	

Annex 3. Chemical data of sediments from the sampling sites in the Weinviertel (Lower Austria). Samples were taken from fall 2009 until fall 2010.
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡IP, inorganic phosphorous; OM, organic matter; OP, organic phosphorous; TP, total phosphorous.

sites †	TP	OP	IP	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Hip	475	202	274	1.86	3.10	0.84	246
Hip	793	329	464	2.55	7.11	1.13	444
Hip	1110	866	244	1.70	3.23	0.77	585
Hip	565	300	264	2.82	7.62	1.84	287
Hip	554	175	379	2.10	7.01	1.81	273
Hip	592	234	358	3.10	10.43	2.20	347.9
Hip	788	288	500	3.54	2.68	1.37	51
Hip	720	285	434	2.87	2.14	1.11	26
Hip	653	29	623	1.28	0.02	0.70	55
Hip	770	390	381	1.71	0.22	1.44	43
Hip	779	429	350	1.61	1.66	1.88	29
Hip	811	456	355	2.07	0.82	1.32	57
Stu I	961	346	615	4.13	4.59	1.47	17
Stu I	965	332	633	3.86	4.16	1.72	9
Stu I	1041	363	678	2.63	3.50	1.19	22
Stu I	1132	440	692	3.79	8.05	1.18	21
Stu I	1361	748	613	2.78	3.96	1.21	21
Stu I	1028	560	468	4.61	0.23	1.69	488
Stu I	1076	569	507	2.32	0.32	1.16	373
Stu I	937	348	590	5.64	0.32	2.15	408
Stu I	1055	454	601	4.65	0.53	2.02	546
Stu I	1316	722	594	4.93	0.13	2.14	554
Stu I	1194	568	626	3.67	0.08	1.10	526
Stu I	1015	542	473	2.86	1.03	0.97	68
Stu I	879	406	472	2.71	0.04	1.02	64
Stu I	992	490	502	3.30	1.18	0.65	63
Stu I	1209	557	652	2.37	0.02	0.68	26
Stu I	1053	270	783	3.08	1.04	0.63	21
Stu I	1122	584	538	2.69	0.61	1.13	23
Stu II	961	454	506	2.12	1.79	2.03	29
Stu II	1004	463	541	2.62	5.58	3.24	25
Stu II	834	344	490	1.55	2.37	1.43	21
Stu II	725	245	481	2.04	2.03	1.96	22
Stu II	744	279	465	1.88	1.89	1.73	20
Stu II	781	346	435	1.20	0.04	0.96	325
Stu II	835	255	580	1.39	0.01	1.11	420
Stu II	887	319	568	2.06	0.09	1.27	342
Stu II	611	319	292	2.30	0.08	1.57	299
Stu II	679	371	308	2.18	0.02	2.28	275
Stu II	486	247	240	3.29	0.06	5.02	201
Stu II	560	334	226	0.86	0.81	0.69	14

sites †	TP	OP	IP	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment	$\mu g\ g^{-1}$ dry sediment
Stu II	481	195	286	1.84	1.34	0.85	10
Stu II	566	262	304	0.83	0.47	0.76	10
Stu II	802	413	388	1.71	0.61	0.89	32
Stu II	729	321	407	1.55	0.91	0.86	38
Stu II	574	277	296	1.05	0.02	0.69	29
Str I	1025	533	492	3.26	0.73	0.94	12
Str I	691	290	401	1.59	0.11	0.46	19
Str I	1037	343	694	2.01	0.26	0.21	15
Str I	956	373	583	1.15	5.31	0.39	52
Str I	823	229	594	1.43	1.70	0.41	26
Str I	773	102	671	2.65	5.32	1.28	41
Str I	1067	332	735	3.19	2.52	0.53	36
Str I	1059	295	764	6.00	5.36	0.34	50
Str I	1139	442	698	1.19	1.01	0.83	52
Str I	731	213	518	2.24	0.24	0.55	7
Str I	927	338	590	2.66	0.05	0.71	10
Str I	1208	412	795	3.25	0.21	1.18	540
Str I	1025	434	590	1.54	0.03	0.41	479
Str I	991	421	570	1.88	0.06	0.44	461
Str I	1177	546	632	3.11	1.75	1.47	474
Str I	1134	524	611	2.90	2.08	1.03	434
Str I	1232	494	738	3.45	2.55	1.46	585
Str I	896	302	593	1.74	0.50	0.79	56
Str I	856	272	585	1.17	0.92	0.47	108
Str I	1051	288	763	2.21	0.74	1.02	69
Str I	898	320	577	1.19	1.09	0.34	55
Str I	1001	252	748	2.84	1.20	0.42	123
Str I	855	297	558	2.00	1.45	0.52	43
Str II	1589	755	834	2.78	2.20	1.01	36
Str II	1746	701	1045	4.40	1.66	1.68	45
Str II	1338	521	817	4.41	0.58	1.57	97
Str II	1352	262	1089	6.20	8.26	0.86	48
Str II	1561	406	1156	6.09	13.65	2.60	153
Str II	1924	309	1615	12.69	14.52	1.37	39
Str II	2055	746	1310	6.57	14.94	1.13	65
Str II	1332	347	986	3.49	8.88	1.47	81
Str II	1598	623	975	5.30	6.59	1.71	73
Str II	1787	658	1129	6.45	6.06	1.62	82
Str II	2089	841	1247	11.01	9.97	3.04	110
Str II	2182	929	1253	8.25	11.40	1.18	68
Str II	1561	691	870	5.13	7.40	1.69	63
Str II	2147	1299	848	3.91	3.80	0.89	54
Str II	1994	927	1067	6.95	1.13	1.09	821
Str II	1548	608	940	3.70	0.61	0.81	623
Str II	2618	1476	1141	7.30	2.86	0.97	1092
Str II	1291	418	873	6.99	10.24	1.44	535
Str II	977	425	552	3.65	4.09	1.00	394

sites †	TP	OP	IP	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Str II	1807	815	992	5.09	0.43	1.50	569
Str II	936	484	452	1.86	0.56	0.65	43
Str II	729	338	392	0.65	0.69	0.40	35
Str II	750	319	432	1.31	0.74	0.48	43
Str II	1509	634	876	4.28	0.55	0.59	109
Str II	1277	444	834	3.53	0.04	0.81	106
Str II	2688	1954	734	1.01	0.24	0.42	59
Hbb I	819	491	329	1.69	4.94	3.41	72
Hbb I	797	524	273	1.58	3.48	3.29	71
Hbb I	855	561	294	1.18	0.69	2.30	96
Hbb I	1678	1164	515	0.94	1.21	0.26	28
Hbb I	2144	1340	804	3.02	3.39	1.96	23
Hbb I	1238	534	704	1.23	2.50	1.35	27
Hbb I	752	528	224	0.86	0.63	1.26	42
Hbb I	883	644	239	0.69	0.12	0.98	74
Hbb I	966	587	379	1.17	1.16	1.28	87
Hbb I	189	-25	215	0.75	2.37	0.95	28
Hbb I	721	399	323	0.43	0.12	0.79	71
Hbb II	1339	432	907	1.78	2.88	0.66	6
Hbb II	2048	1426	622	5.31	3.70	1.81	5
Hbb II	1352	822	530	1.69	1.41	0.89	11
Hbb II	852	424	428	0.58	0.11	0.36	33
Hbb II	1059	543	516	0.70	0.04	0.55	69
Hbb II	806	379	0.35	0.26	0.13	0.26	15
Hbb II	1323	778	0.70	0.46	0.31	0.46	33
Hbb II	720	357	0.93	0.85	0.04	0.85	12
Hbg I	951	367	2.15	1.79	0.47	1.79	35
Hbg I	1045	381	2.52	2.46	1.20	2.46	30
Hbg I	1072	508	563	1.50	0.60	1.27	19
Hbg I	1370	621	749	1.76	1.55	1.77	69
Hbg I	1216	140	1076	2.17	1.58	1.80	88
Hbg I	1233	417	816	0.66	0.19	1.28	24
Hbg I	1189	309	880	2.01	2.45	1.49	49
Hbg I	947	129	818	1.62	2.53	1.99	36
Hbg I	1149	479	670	3.55	0.03	1.27	121
Hbg I	1116	424	691	2.20	0.03	0.52	63
Hbg I	1515	577	938	5.39	0.03	1.39	217
Hbg I	1115	345	770	4.44	0.03	0.84	68
Hbg I	1269	537	732	2.30	0.18	0.45	146
Hbg II	1145	508	638	3.90	1.61	1.56	20
Hbg II	1214	628	586	4.15	2.17	2.20	24
Hbg II	1040	502	538	4.31	1.39	2.36	21
Hbg II	1114	497	617	1.90	3.93	1.60	17
Hbg II	760	263	497	0.83	0.11	1.10	18
Hbg II	1000	431	569	1.68	1.74	1.50	18
Hbg II	721	70	651	0.47	-0.03	0.60	18
Hbg II	1590	191	1399	6.69	2.26	1.80	16

sites †	TP	OP	IP	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Hbg II	757	336	422	2.61	0.03	0.96	12
Hbg II	807	338	470	2.71	0.03	1.14	18
Hbg II	893	381	513	1.57	0.09	0.68	22
Hbg II	974	416	558	4.90	0.08	1.12	11
Hbg II	955	475	480	5.77	0.03	2.21	11

Annex 4. Structural and chemical data of sediments from the sampling sites in the Weinviertel (Lower Austria).
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡IP, inorganic phosphorous; OM, organic matter; OP, organic phosphorous; TP, total phosphorous.

sites †	depth of oxic layer	mean grain size	OM	C:N	N:P	$\delta^{13}C$
	10^{-2} m	10^{-6} m	%			‰
Hip	7	61.5	3.3	7.23	4.27	-26.97
Hip	15	57.2	4.5	6.84	1.93	-27.16
Hip	8	273.0	2.7	6.82	1.74	-27.12
Hip	12	212.8	3.7	6.07	1.86	-27.19
Hip	7	92.9	3.7	6.07	2.62	-26.98
Hip	13	52.8	9.7	7.66	1.87	-26.55
Hip		49.2	5.0	5.60	2.41	-26.92
Hip		68.8	4.4	6.67	2.31	-27.33
Hip		47.2	4.9	6.41	1.24	-27.01
Hip		40.9	6.1	6.99	2.63	-27.49
Hip		37.6	6.0	6.51	1.39	-26.04
Hip		39.1	6.2	10.29	3.49	-27.34
Hip				10.68		-27.32
Hip				10.21		-27.87
Hip				9.08		-27.27
Hip				9.38		-27.36
Hip				9.12		-27.44
Hip				6.55		-26.24
Stu I	1	41.9	6.1	9.07	2.51	-27.70
Stu I	3	36.9	4.9	9.20	2.59	-27.63
Stu I	4	36.1	5.3	10.08	2.69	-27.64
Stu I	10	34.5	3.5	9.03	2.57	-26.82
Stu I	9	34.7	7.0	11.46	2.63	-26.99
Stu I	7	35.0	5.1	12.84	3.09	-27.31
Stu I	3	38.0	5.1	11.86	2.81	-27.94
Stu I	4	38.5	5.6	10.94	3.61	-27.92
Stu I	5	34.9	5.0	13.41	3.01	-27.07
Stu I	6	34.5	6.2	8.02	2.17	-27.53
Stu I	3	35.8	6.3	7.67	1.76	-27.02
Stu I	5	33.9	6.0	11.10	2.46	-28.86
Stu I				8.84	1.28	-27.09
Stu I				8.19	2.77	-26.78
Stu I				8.20	2.32	-26.14
Stu I				8.75	1.90	-26.84
Stu I				8.35	1.90	-26.63
Stu I				8.34		-26.39
Stu II	4	43.6	3.9	12.61	3.63	-26.71
Stu II	3	48.5	3.4	8.64	1.18	-28.93
Stu II	4	45.8	4.1	13.18	3.19	-27.44
Stu II	2	0.0	2.0	11.28	3.46	-27.49
Stu II	4	0.0	3.4	10.28	2.52	-27.30
Stu II	3	0.0	1.4	10.85	4.37	-27.76

sites †	depth of oxic layer	mean grain size	OM	C:N	N:P	$\delta^{13}C$
	10^{-2} m	10^{-6} m	%			‰
Stu II	2	41.6	8.6	9.81	2.14	-27.82
Stu II	4	50.3	6.9	9.60	1.90	-27.60
Stu II	4	50.3	9.7	10.01	2.97	-28.16
Stu II	3	51.2	4.0	11.15	4.09	-28.32
Stu II	6	43.6	4.0	11.27	7.36	-28.15
Stu II	6	46.2	4.6	10.50	3.85	-25.80
Stu II				9.27	3.66	-27.61
Stu II				6.32	3.34	-26.81
Stu II				4.86	1.69	-26.36
Stu II				8.43	0.78	-25.95
Stu II				7.70	1.39	-26.63
Stu II				5.78		-27.38
Stu II				10.26		-27.56
Stu II				9.37		-27.68
Stu II				10.01		
Str I	5	47.5	3.0	11.46	3.77	-29.02
Str I	1	141.7	4.2	12.78	5.73	-28.56
Str I	5	81.7	4.0	8.85	3.12	-27.83
Str I	8	60.7	5.0	7.21	3.38	-28.42
Str I	7	61.5	4.8	8.65	2.77	-28.64
Str I	3	54.5	7.1	7.72	3.03	-28.53
Str I		85.4	3.8	5.74	0.79	-26.24
Str I		75.9	3.9	4.97	0.66	-25.87
Str I		47.2	5.2	5.53	0.74	-26.29
Str I		49.8	5.7	7.84	3.32	-26.64
Str I		39.9	5.4	8.93	3.58	-27.05
Str I		48.0	5.0	7.68	1.85	-26.54
Str I				8.02	2.29	-26.64
Str I				7.32	1.67	-25.66
Str I				7.65	2.46	-26.91
Str I				6.82	1.82	-26.29
Str I				7.42	1.36	-26.79
Str I				7.20	1.41	-26.18
Str II	7	40.4	6.9	9.16	2.83	-27.96
Str II	7	36.7	8.0	9.44	2.65	-28.10
Str II	4	39.6	9.9	9.42	3.26	-27.87
Str II	4	39.1	6.2	12.00	3.81	-28.23
Str II	4	38.8	6.9	11.61	3.30	-27.94
Str II	4	37.8	8.8	12.66	2.59	-27.77
Str II		36.4	8.5	12.73	2.78	-27.79
Str II		40.5	9.0	10.41	3.50	-27.62
Str II		42.6	10.6	11.14	3.40	-27.88
Str II		35.4	8.9	9.30	2.06	-27.39
Str II		34.9	8.0	8.97	1.62	-27.49
Str II		36.6	8.9	10.61	1.52	-27.87
Str II				9.27	2.53	-27.53
Str II				9.86	1.94	-27.72

sites †	depth of oxic layer	mean grain size	OM	C:N	N:P	$\delta^{13}C$
	10^{-2} m	10^{-6} m	%			‰
Str II				9.39	1.95	-27.31
Str II				8.62	1.70	-27.36
Str II				8.67	1.42	-27.13
Str II				8.25	2.72	-27.31
Hbb I	15	44.5	16.3			
Hbb I	15	42.6	10.7			
Hbb I	4	40.4	11.6			
Hbb I		39.1	10.8			
Hbb I		41.6	11.6			
Hbb II	3	35.8	7.0			
Hbb II	4	39.4	12.9			
Hbb II	0.5	35.5	5.5			
Hbb II		36.3	8.2			
Hbb II		36.9	9.2			
Hbg I	3	33.4	5.6			
Hbg I	4	33.2	5.1			
Hbg I	10	33.4	6.2			
Hbg I	1	33.1	5.3			
Hbg I	0.5	33.1	5.6			
Hbg I	8					
Hbg I	6					
Hbg I	7					
Hbg I	9					
Hbg I	8					
Hbg II	10	47.6	3.5			
Hbg II	7	36.1	5.9			
Hbg II	7	36.8	5.8			
Hbg II	10	37.2	4.1			
Hbg II	10	36.4	4.8			
Hbg II	5					
Hbg II	10					
Hbg II	12					
Hbg II	4					
Hbg II	4					

Annex 5. Structure and chemical data of the upper sediment layers from the sampling sites in the Weinviertel (Lower Austria).
95 samples were taken from fall 2009 until fall 2010.
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡IP, inorganic phosphorous; OM, organic matter; OP, organic phosphorous; TP, total phosphorous.

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Hip	35.7	4.5	547	209	338	6.20	3.94	3.55	286
Hip	35.9	0.8	427	228	199	1.73	2.03	0.57	149
Hip	232.0	4.8	1002	630	372	5.86	0.99	3.23	658
Hip	79.6	2.5	305	104	201	3.12	3.37	1.06	170
Hip	326.3	4.1	787	357	430	2.84	7.29	0.95	362
Hip	228.6	1.6	1919	1724	195	2.78	2.86	0.45	925
Hip	289.5	3.1	415	98	317	2.96	8.14	0.86	212
Hip	59.0	2.0	437	197	240	1.76	1.08	1.07	215
Hip	52.8	4.0	637	299	338	2.97	3.33	0.87	348
Hip	2858.6	6.5	956	290	666	3.88	1.66	1.27	28
Hip	44.3	3.7	540	43	498	3.54	1.43	1.43	17
Hip	48.3	6.3	1039	234	805	2.81	0.03	1.55	25
Hip	51.7	5.7	818	373	445	3.79	4.81	1.15	30
Hip	56.4	7.4	802	283	520	4.35	3.77	1.95	31
Hip	43.5	7.3	739	-1100	1838	2.28	0.03	1.37	22
Hip	39.4	5.2	794	360	434	2.29	0.76	1.80	39
Hip	39.2	5.9	800	402	399	2.23	3.13	2.39	24
Hip		6.1	901	475	426	2.94	1.36	1.42	66
Stu I	35.3	5.6	894	341	553	4.78	11.18	1.79	24
Stu I	43.0	5.7	1140	531	609	6.21	5.77	3.66	27
Stu I	38.7	4.9	1056	477	579	3.21	2.60	1.46	47
Stu I	34.3	0.8	1037	447	590	3.81	10.27	1.53	39
Stu I	35.6	10.7	1806	1321	484	2.94	7.12	1.27	31
Stu I	35.5	5.4	1088	605	482	7.42	0.39	2.18	532

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Stu I	40.4	5.1	1071	607	464	0.60	0.38	0.32	358
Stu I	36.4	4.3	895	360	535	5.27	0.14	2.17	399
Stu I	36.2	6.8	976	417	558	5.56	0.44	2.79	546
Stu I	34.9	5.4	1336	664	672	3.04	0.04	0.98	542
Stu I	35.3	7.0	1230	626	604	2.73	0.10	0.63	515
Stu I	35.3	6.2	1085	534	551	3.66	0.03	1.36	154
Stu I			974	459	515	2.18	0.06	1.00	119
Stu I			1057	511	547	3.71	1.31	0.54	74
Stu I			1158	485	673	1.80	0.02	0.75	19
Stu I			1154	526	627	3.53	1.90	1.04	21
Stu I			1197	608	589	2.63	0.03	1.28	16
Stu II	49.9	4.2	1042	489	554	2.69	3.81	2.92	32
Stu II	61.8	2.8	1101	564	537	3.02	9.33	3.36	30
Stu II	55.5	3.8	1035	511	525	2.05	6.43	2.39	25
Stu II	53.1	1.5	787	298	488	1.03	4.83	1.73	27
Stu II	203.4	2.9	787	349	438	2.05	4.00	2.14	18
Stu II	82.2	1.2	810	362	448	1.17	0.07	1.10	349
Stu II	38.2	6.8	837	113	724	1.25	0.01	1.05	450
Stu II	36.0	5.1	934	455	479	2.42	0.16	1.29	383
Stu II	38.9	6.7	597	395	202	1.85	0.11	0.51	277
Stu II		5.9	670	319	351	2.28	0.03	2.36	280
Stu II		3.8	242	106	136	2.92	0.19	0.71	119
Stu II		5.1	538	326	212	0.89	1.16	0.72	14
Stu II			476	169	307	2.16	1.94	0.71	9
Stu II			515	148	367	0.90	1.13	0.74	8
Stu II			857	511	346	1.80	1.59	0.96	21
Stu II			747	310	437	1.89	1.77	1.19	40
Stu II			763	385	378	1.16	0.03	1.02	25
Str I	51.4	2.8	1119	425	694	2.53	14.81	0.90	129

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Str I	263.1	3.9	944	319	625	1.62	3.81	0.59	35
Str I	297.8	4.4	922	132	791	4.82	12.99	2.55	89
Str I	132.9	4.4	1540	620	920	7.74	7.24	0.45	37
Str I	57.8	5.9	1586	654	932	14.34	11.97	0.40	78
Str I	87.5	4.4	1286	545	741	0.82	1.77	0.96	44
Str I	58.5	4.6	680	155	524	1.34	0.49	0.35	4
Str I	82.5	3.5	999	497	502	1.68	0.09	0.34	3
Str I	52.3	4.9	1074	116	958	1.30	0.67	0.50	513
Str I	44.6	5.9	889	365	524	1.35	0.03	0.58	345
Str I	40.0	6.1	968	417	551	1.71	0.12	0.49	475
Str I	141.9	4.2	972	307	665	4.57	9.70	0.90	440
Str I			1134	424	710	3.02	7.68	1.22	555
Str I			1210	376	835	4.80	11.52	1.17	524
Str I			870	220	650	2.13	1.07	1.05	17
Str I			1175	332	844	1.87	0.05	0.88	124
Str I			1129	244	885	2.32	0.48	1.54	38
Str I			844	303	541	1.94	0.04	0.72	30
Str I			2092	311	1781	7.19	1.74	1.40	72
Str I			950	384	566	2.21	2.17	0.75	30
Str II	46.0	6.4	1415	334	1081	8.11	12.56	0.81	47
Str II	36.8	9.1	1960	716	1245	13.84	37.42	6.60	346
Str II	45.0	11.0	2108	504	1605	14.94	7.69	1.03	25
Str II	36.8	4.7	2141	883	1259	11.92	34.22	2.30	112
Str II	37.9	5.0	1696	565	1131	7.40	27.86	4.15	197
Str II	36.4	9.0	2039	888	1151	8.70	11.96	1.56	87
Str II	36.9	8.1	1781	681	1100	8.52	10.15	0.75	72
Str II	42.3	7.8	2070	869	1202	10.78	11.02	1.99	105
Str II	44.8	9.0	2059	906	1153	12.72	19.86	1.45	70
Str II	35.3	9.1	1852	932	919	8.30	13.67	1.78	91

sites †	mean grain size	OM	TP	OP	IP ‡	$P\text{-}PO_4$	$N\text{-}NO_3$	$N\text{-}NO_2$	$N\text{-}NH_4$
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Str II	33.9	7.5	1174	369	804	5.52	6.89	0.92	67
Str II	37.2	8.5	2598	1323	1276	11.07	2.27	1.22	980
Str II			2137	812	1325	11.18	2.80	1.30	744
Str II			4243	2606	1637	13.73	6.42	1.44	1737
Str II			1344	383	961	8.86	16.55	1.37	540
Str II			1507	579	928	9.25	19.80	1.64	767
Str II			2095	960	1135	6.78	0.81	2.01	529
Str II			1133	620	513	2.45	1.14	0.72	30
Str II			1316	692	624	0.86	1.36	0.64	40
Str II			900	346	553	1.75	1.34	0.58	38
Str II			1810	735	1075	5.88	0.80	0.69	120
Str II			1786	640	1146	5.19	0.07	1.33	135
Str II			1602	657	945	1.74	0.31	0.48	47

Annex 6. Structure and chemical data of the lower sediment layers from the sampling sites in the Weinviertel (Lower Austria).
95 samples were taken from fall 2009 until fall 2010.
† Hip, Hipples stream; Str, Stronsdorf stream; Stu, Stuetzenhofen stream; Hbb, Herbertsbrunn stream; Hbg, Herrnbaumgarten stream.
‡IP, inorganic phosphorous; OM, organic matter; OP, organic phosphorous; TP, total phosphorous.

sites †	mean grain size	OM	TP	OP	IP ‡	$P\text{-}PO_4$	$N\text{-}NO_3$	$N\text{-}NO_2$	$N\text{-}NH_4$
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Hip	34.5	4.2	524	207	318	6.03	1.25	3.25	206
Hip	34.2	3.3	540	334	206	0.76	0.32	0.85	219
Hip	51.6	2.8	1471	348	1123	7.43	0.05	3.09	858
Hip	53.5	3.7	561	251	310	1.23	2.96	0.73	284
Hip	183.1	4.6	797	313	483	2.39	7.00	1.23	490
Hip	205.9	3.4	578	301	277	0.98	3.47	0.98	361
Hip	57.1	4.1	625	383	243	2.77	7.41	2.25	318
Hip	50.5	4.3	605	166	439	2.24	9.57	2.13	297
Hip	42.0	12.2	572	206	366	3.16	13.55	2.79	348
Hip	164.8	6.4	737	367	369	2.41	0.03	1.79	41
Hip	39.8	2.8	426	132	293	0.67	0.02	0.51	9
Hip	468.6	6.3	735	382	354	2.88	0.03	1.49	54
Hip	114.9	4.7	843	273	570	3.73	1.91	1.60	66
Hip	45.3	3.6	750	307	443	2.69	1.87	0.97	26
Hip	39.8	4.3	686	320	366	1.14	0.02	0.58	68
Hip	35.8	6.5	888	467	421	1.75	0.03	1.52	51
Hip	39.0	6.2	914	557	357	1.10	0.03	1.57	43
Hip		6.2	969	578	391	1.80	0.51	1.63	65
Stu I	51.0	6.6	998	349	649	3.76	0.93	1.30	14
Stu I	35.3	4.7	902	260	642	3.01	3.57	1.02	3
Stu I	34.7	5.6	1033	303	729	2.33	3.98	1.05	9
Stu I	34.6	4.7	1190	436	754	3.78	6.70	0.96	11
Stu I	34.4	5.7	1106	418	687	2.68	2.14	1.18	16
Stu I	48.7	4.9	956	505	451	1.21	0.03	1.08	435

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	μg g^{-1} dry sediment	μg g^{-1} dry sediment	μg g^{-1} dry sediment	μg g^{-1} dry sediment	μg g^{-1} dry sediment	μg g^{-1} dry sediment	μg g^{-1} dry sediment
Stu I	36.7	5.1	1080	533	547	3.92	0.25	1.94	386
Stu I	42.4	7.6	977	335	642	6.00	0.50	2.14	417
Stu I	34.5	4.3	1134	490	644	3.75	0.62	1.26	546
Stu I	34.4	6.5	1297	778	519	6.74	0.23	3.26	565
Stu I	36.0	6.0	1159	511	648	4.58	0.06	1.56	536
Stu I	33.4	5.8	1135	623	512	2.97	1.56	0.96	46
Stu I			969	442	527	3.59	0.03	1.23	36
Stu I			1053	542	511	3.14	1.14	1.00	54
Stu I			1369	650	719	2.87	0.03	0.73	33
Stu I			1093	183	910	3.14	0.76	0.50	22
Stu I			1207	634	573	2.99	0.89	1.19	28
Stu II	41.9	3.8	911	433	478	1.77	0.56	1.49	27
Stu II	38.7	4.0	942	399	543	2.37	3.19	3.17	22
Stu II	40.7	4.3	719	248	471	1.26	0.06	0.88	19
Stu II	37.6	2.6	683	208	475	2.73	0.10	2.11	18
Stu II	38.2	3.8	707	219	488	1.74	0.08	1.38	21
Stu II	40.2	1.7	747	328	419	1.25	0.00	0.81	298
Stu II	60.6	9.5	832	421	411	1.57	0.00	1.17	384
Stu II	46.8	8.6	823	137	686	1.58	0.00	1.23	287
Stu II	51.4	13.1	623	252	371	2.70	0.06	2.50	318
Stu II		3.1	689	424	265	2.08	0.00	2.21	269
Stu II		4.1	606	315	290	3.48	0.00	7.12	241
Stu II		4.3	764	441	323	0.99	0.03	0.79	16
Stu II			595	305	290	1.36	0.03	1.38	14
Stu II			681	385	296	0.88	0.02	0.88	12
Stu II			862	397	465	1.85	0.03	0.96	44
Stu II			799	375	424	1.36	0.03	0.58	41
Stu II			627	294	333	1.29	0.03	0.71	40
Str I	43.4	3.1	866	345	522	0.39	0.04	0.10	8

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Str I	62.8	4.4	733	162	571	1.30	0.14	0.28	19
Str I	49.7	3.6	689	85	603	1.42	0.96	0.55	14
Str I	57.3	5.1	860	206	654	1.20	0.45	0.56	35
Str I	62.9	4.4	727	69	658	0.74	1.20	0.29	32
Str I	51.8	7.8	957	314	643	1.66	0.06	0.68	62
Str I	125.8	3.2	778	266	512	3.08	0.02	0.74	10
Str I	71.4	4.3	858	184	674	3.60	0.02	1.07	16
Str I	45.0	5.4	1262	532	729	4.04	0.02	1.45	551
Str I	138.9	5.0	1103	474	629	1.64	0.03	0.32	556
Str I	39.7	4.7	1007	423	583	1.99	0.02	0.41	452
Str I	40.6	5.4	1222	598	624	2.80	0.02	1.60	481
Str I			1135	561	574	2.85	0.02	0.97	389
Str I			1238	527	711	3.07	0.03	1.54	602
Str I			1030	416	614	1.60	0.03	0.64	100
Str I			936	309	627	1.21	1.36	0.44	125
Str I			1151	391	760	2.49	1.17	0.60	118
Str I			1003	357	646	0.98	1.63	0.21	71
Str I			842	262	580	2.11	1.18	0.23	147
Str I			919	276	644	2.17	1.10	0.41	60
Str II	38.4	7.5	1270	170	1100	3.75	2.73	0.91	59
Str II	36.5	7.8	1362	250	1111	2.21	1.74	0.61	55
Str II	37.0	8.9	1761	137	1624	10.70	20.57	1.67	52
Str II	41.3	8.4	1990	642	1348	2.51	0.31	0.23	30
Str II	39.3	7.4	1170	250	920	1.75	0.40	0.27	29
Str II	39.0	8.6	1205	387	818	2.27	1.80	1.85	60
Str II	35.6	9.2	1791	642	1149	5.03	3.24	2.22	88
Str II	38.1	10.9	2103	821	1282	11.18	9.17	3.85	114
Str II	41.6	11.4	2330	956	1374	2.87	1.18	0.85	66
Str II	35.4	8.8	1273	452	821	1.98	1.16	1.60	36

sites †	mean grain size	OM	TP	OP	IP ‡	P-PO_4	N-NO_3	N-NO_2	N-NH_4
	10^{-6} m	%	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment	µg g^{-1} dry sediment
Str II	37.4	9.2	3181	2286	895	2.19	0.51	0.86	40
Str II	35.8	9.5	1418	550	868	3.00	0.04	0.97	670
Str II			1393	554	838	1.74	0.04	0.68	592
Str II			1331	582	749	2.21	0.03	0.60	582
Str II			1205	474	731	3.97	0.03	1.56	527
Str II			840	386	454	2.21	0.02	0.83	297
Str II			1498	661	838	3.27	0.04	0.96	612
Str II			904	430	474	1.58	0.03	0.70	67
Str II			363	96	267	0.59	0.25	0.26	37
Str II			727	341	386	1.11	0.30	0.46	54
Str II			1419	661	758	2.11	0.15	0.58	131
Str II			1156	387	769	3.02	0.03	0.60	106
Str II			4289	3554	736	0.64	0.25	0.47	84

CURRICULUM VITAE

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