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Fine Sediment Removal Influences Biogeochemical Processes in a Gravel-bottomed Stream

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Abstract

The transport and processing of nutrients and organic matter in streams are important functions that influence the condition of watersheds and downstream ecosystems. In this study, we investigated the effects of streambed sediment removal on biogeochemical cycling in Fawn River, a gravel-bed river in Indiana, U.S.A. We measured stream metabolism as well as nitrogen (N) and phosphorus (P) retention in both restored and unrestored reaches of Fawn River to examine how sediment removal affected multiple biogeochemical functions at the reach scale. We also assessed the properties of restored and unrestored streambed sediments to elucidate potential mechanisms driving observed reach-scale differences. We found that sediment removal led to lower rates of primary productivity and ecosystem respiration in the restored reach, likely due to macrophyte removal and potentially changes to sediment organic matter quality. We found minimal differences in N and P retention, suggesting that these processes are controlled at larger spatial or temporal scales than were examined in this study. Denitrification enzyme activity was lower in sediments from the restored reach compared to the unrestored reach, suggesting that restoration may have decreased N removal. Our results indicate that most near-term changes in biogeochemical function following restoration could be attributed to macrophyte removal, although effects from sediment removal may emerge over longer timescales.

Keywords

Stream restoration; Nutrient retention; Ecosystem metabolism; Fine sediment Macrophyte removal; Fawn River

Introduction

Background

Human activities contribute to the impairment of surface waters through nutrient loading (Carpenter et al. 1998), habitat alteration, and sediment input (Waters 1995). Nutrient and organic matter (OM) loading can cause undesirable conditions, such as hypoxic zones (e.g. Paerl et al. 1998) and harmful algal blooms (Anderson et al. 2002), which can impair

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aesthetic, biological, and commercially important ecosystem values. Nutrient and OM loading from the landscape is a primary controlling factor on nitrogen (N), phosphorus (P), and OM concentrations in streams. However, biogeochemical processing in streams also influences the magnitude and timing of the delivery of nutrients and OM to downstream ecosystems (e.g. Mulholland et al. 2008). Two commonly assessed stream biogeochemical functions are metabolism and nutrient retention. Stream metabolism describes the production and removal of OM in terms of gross primary production (GPP) and ecosystem respiration (ER) and can be estimated from diel changes in dissolved oxygen (DO) concentration (Hall 2016). Nutrient retention represents the sum of biotic and abiotic processes that control the storage and transformation of N and P, and can be estimated in streams using the nutrient spiraling approach, which quantifies transport and retention, often as part of a controlled nutrient release experiment (Ensign and Doyle 2006).

Metabolism and nutrient retention are influenced by stream morphological characteristics, such as streambed material, habitat quality, transient storage, and longitudinal profile (e.g. Fellows et al. 2001, Gücker and Boëchat 2004). In-stream vegetation and streambed sedimentation are commonly associated with impairment from human activities (e.g., Collins and Walling 2007) and can negatively impact streambed composition, habitat quality, and surface–subsurface exchange. Submerged and/or emergent macrophytes can have an important role in mediating instream nutrient retention and metabolism through increasing autotrophic N and P demand, GPP, and respiration by macrophyte roots (Balestrini et al. 2018). Furthermore, macrophytes may indirectly influence heterotrophic metabolism and nutrient demand through effects on stream hydraulics (Dodds and Biggs 2002) and effects on microbial activity in macrophyte-associated sediments (Forshay and Dodson 2011). Fine benthic sediments affect metabolism and nutrient retention by influencing habitat characteristics and faunal communities, impairing connectivity with the hyporheic zone (Mulholland et al. 2001), and altering adsorption of nutrient molecules to sediment particles (Lottig and Stanley 2007).

Human-induced morphological changes in stream channels, such as straightening, dredging, and sedimentation may have consequences for stream biogeochemical functions by altering the distribution of vegetation and fine sediments. Stream restoration has also emerged as a widespread source of stream channel alteration, as restoration projects intentionally alter stream morphology to improve channel stability, aesthetics, water quality, or ecological functioning (Bernhardt et al. 2005, Palmer et al. 2014). Researchers and policymakers have also begun to explore the use of strategically planned stream restoration techniques that improve hyporheic connectivity or restore natural flooding regimes as an approach for managing water quality in downstream ecosystems (Craig et al. 2008). As stream restoration becomes increasingly widespread, it is crucial to assess the efficacy of the various practices intended to influence stream biogeochemical functions (Palmer et al. 2005, Newcomer-Johnson et al. 2016), as well as potential unintended biogeochemical consequences of stream restorations not specifically targeting water quality (Bukaveckas 2007). Studies examining multiple biogeochemical functions simultaneously may help determine how different biogeochemical functions emerge to varying extents and over varying timescales following restoration.

In this study, we examined biogeochemical responses in Fawn River, Indiana (USA), to restoration involving fine sediment removal (sediment release and restoration are described below). The Fawn River restoration provided an opportunity to examine the effects of restoration on stream metabolism and nutrient spiraling in a real-world setting in which a limited number of physical and chemical variables—streambed sediment content and macrophyte abundance—were manipulated in a semi-controlled manner. In a companion study, Ward et al. (2018) found that fine sediment removal likely increased surface water exchange with the hyporheic zone in Fawn River during high-flow periods. Increased surface–subsurface connectivity is expected to promote ecological functioning, including nutrient retention (Mendoza-Lera and Datry 2017). However, anaerobic processes, such as denitrification, and nutrient retention in general, are often enhanced by accumulations of OM within stream channels (Valett et al. 2002, Newcomer et al. 2012).

We hypothesized that (1) GPP would be reduced at restored sites due to the removal of macrophytes, (2) ER would be reduced concurrently with primary productivity due to elimination of macrophyte respiration and reduction in bioavailable autochthonous organic matter, (3) N retention would be reduced due to both the reduction of autotrophic N demand and the reduction in denitrification following reconnection of sediment interstices with O₂-rich surface water, and (4) P retention would be reduced due to the reduction of autotrophic P demand and removal of fine benthic sediments that support P adsorption.

Methods

Site Description

We evaluated the effects of restoration involving sediment removal on nutrient retention and stream metabolism in Fawn River, a low-gradient stream within the St. Joseph River watershed in southern Michigan and northern Indiana. Fawn River flows from east to west, draining a mixed land use, 240 km² catchment (Wesley 2000). The 8-km section of Fawn River bounded by the Fawn River Fish Hatchery on the upstream end (near Orland, IN) and a mill pond on the downstream end (near Greenfield Mills, IN) is known as the lower Fawn River (Fig. 1). The lower Fawn River was a unique biological resource that retained many geomorphic and ecological features of a largely undisturbed, gravel-bottom stream (Lindsey et al. 1970). The geomorphology and ecology of the lower Fawn River was severely impaired in 1998 by the catastrophic release of nearly 100,000 m³ of unconsolidated reservoir silt from the fish hatchery. These sediments travelled downstream as a hyper-concentrated flow, extending across the entire length of the lower Fawn River until they were stopped by the impoundment at Greenfield Mills ~4 river km downstream from our study area. In addition to the significant mortality of wildlife from the initial disturbance of the sediment release, the natural gravel substrate of Fawn River was covered with a layer of fine sediment ranging from 10–100 cm in thickness. This finer substrate facilitated the colonization of submerged macrophyte species, predominantly pondweeds (*Potamogeton* spp.) (Lewis et al. 2013).

Restoration of the impacted reaches of Fawn River began in 2011. This restoration focused on the removal of fine sediment from the natural gravel substrate. Submerged macrophytes were first removed manually from the thalweg to destabilize non-natural sediment deposits.

Fine sediments were then removed from the natural coarse benthic substrate through use of Sand Wand technology (Sepulveda et al. 2014), which suspends fine sediments with a high-pressure jet of water and then pumps the slurry to a dewatering pit. This method does not require heavy equipment and involves minimal disturbance to the riparian vegetation. Subsequent to sediment removal, woody debris structures were installed to focus flow in the thalweg and concentrate deposition along the banks of the river, and also to buffer against future possible sediment releases by stabilizing natural depositional areas.

The engineering firm performing the restoration analyzed sediments before and after restoration at a representative reach of stream. The average pre-restoration particle size distribution (based on mass) shifted from 31.8% sand, 2.8% silt, and 65.5% gravel to a post-restoration distribution of 5.0% sand, 0.1% silt, and 94.9% gravel (Fawn River Restoration and Conservation Charitable Trust, unpublished data). This shift in particle size distribution was clearly visible from in-situ photos of the streambed (Fig. 1 and Ward et al. 2018).

Study Design

Sampling was conducted in a restored reach and an unrestored reach of Fawn River (Fig. 1) during four time periods between June and November of 2013. When evaluating the ecological outcomes of stream restoration, the logistics of restoration activities can constrain the study design in ways that reduce statistical power. In the case of Fawn River, the timing and location of restoration activities was determined by an engineering firm, and restoration was underway when this study was initiated. As a result, we were limited in our ability to obtain truly independent replicate measures of stream function or to conduct a full before-after-control-impact (BACI) design. Therefore, we relied on a space-for-time substitution design (Downes et al. 2002) and acknowledge that the study is pseudo-replicated at the reach scale (i.e., one restored reach and one unrestored reach). Many similarly designed studies have proved informative and ecosystem-scale manipulations are recognized as special cases in which the lack of replication does not preclude insight into ecological patterns and processes (Hurlbert 2004). Nonetheless, caution should be used if extrapolating results from Fawn River to other streams.

Restored and unrestored reaches were identified within the sediment-impacted portion of Fawn River, with the restored reach ~1.5 km upstream of the unrestored reach. Over the course of the study, discharge was typically higher in the unrestored reach than the restored reach (Table 1), although discharge was slightly higher in the restored reach during June. The unrestored reach was deeper and wider, resulting in a water velocity 30–61% lower than the restored reach. Both reaches were losing systems under high-flow conditions, although this trend was more pronounced and persistent in the unrestored reach.

For whole-stream metabolism and measurements based on solute releases (described below) it is preferable to use fixed travel times rather than fixed reach lengths (e.g., Schmadel et al. 2016). Therefore, only the upstream boundaries of the restored and unrestored study reach were fixed. The downstream boundaries were established during each field campaign by releasing rhodamine WT dye at the upstream boundary and determining the longitudinal reach length that corresponded to a 20-min travel time. This approach has the advantage of

standardizing advective timescales across field campaigns and between the restored and unrestored reaches, even though reach lengths varied.

Background Chemistry

Samples for NO_3^- -N, NH_4^+ -N, total N, soluble reactive P (SRP), total P, and dissolved organic carbon (DOC) were collected from the upstream and downstream ends of each study reach daily over each 4- or 5-day field campaign. Samples for total N and total P were unfiltered. Samples for dissolved nutrients were filtered through a 0.45 μm membrane, stored on ice for transport, and kept frozen until analysis. Samples for DOC were filtered and acidified in the field to $\text{pH} = 2$. Prior to analysis, total N was oxidized to nitrate-N using the alkaline persulfate method (Cabrera and Beare 1993) using blanks and a thiamine recovery standard; similarly, total P was oxidized to SRP using an acidic persulfate digestion (Gales et al. 1966) using blanks and an internal recovery standard. Nutrient concentrations were determined using a Lachat Quick-Chem 8500 Flow Injection Analysis system (Hach Company, Loveland, CO), and DOC was determined using a Shimadzu TOC-V CPN (Columbia, MD). For all chemical analyses, commercially certified standards, field duplicates, and blanks were analyzed as part of a quality control protocol.

Sediment Analysis

Sediments were sampled on the last day of each field campaign from the upstream and downstream ends of each study reach. A mid-reach transect was also sampled during the September and November campaigns. The upper 5 cm of benthic sediments were cored five times at 1 m longitudinal intervals, sieved to 2 mm grain size and composited for left, right and center positions along each transect. Sediments not used for measurement of sediment oxygen demand were kept on ice until return to the laboratory, where subsamples were taken for elemental analysis and frozen at -20°C . Phosphate adsorption equilibria and denitrification enzyme activity were measured within 48 h of sediment collection; these sediments were kept refrigerated until analysis. Sediments were dried at 60°C for three days before they were ground to 40 mesh (0.42 mm nominal diameter) using a Wiley mill. Sediments were analyzed for total C content using a Perkin Elmer 2400 CHN analyzer (Waltham, MA, USA; MDL = 0.001 mg) after a 3-day HCl fumigation to remove carbonate-bearing minerals.

Phosphate equilibria were measured according to the method of Beckett and White (1964), using equilibrations with prepared stock solutions containing 0–25,000 mg phosphate-P/L. For each sediment sample, 40 mL of each stock solution was combined with 6 g of wet sediment in a 50-mL centrifuge tube and shaken for 12 h to equilibrate adsorbed and aqueous phases. Tubes were then centrifuged for 15 min at 750 g and the supernatant was filtered (0.45 μm nitrocellulose) into 15 mL centrifuge tubes and frozen until analysis. The SRP concentration of the supernatant was determined as above and represented the equilibrium phosphorus concentration (EPC). This was regressed against the mass of P adsorbed or released per kg of sediment using quadratic regression. The x-intercept of this equation is equal to EPC_0 , the aqueous SRP concentration at which sediments are neither a source nor a sink for P (Fig. 2a, b). When stream water SRP concentrations exceed the

EPC₀, sediments act as a sink for P. When SRP concentrations are less than the EPC₀, sediments can be a source of P to the water column.

Determinations of sediment oxygen demand (SOD) were performed in the field immediately upon sediment collection according to U.S. EPA protocols (e.g., Hill et al. 2000), incubating known amounts of sediment with stream water for 2 h and measuring the reduction in DO. To determine the ash-free dry mass (AFDM) of the sediment, samples were dried at 60 °C to a constant weight and then combusted at 550 °C for 3 h. The material remaining after combustion was rewetted, dried, and reweighed. The difference in mass before and after combustion is the AFDM and represents the organic content of the sediment. The change in DO was divided by the incubation time and AFDM to give SOD in units of mg O₂/g AFDM/h.

Measurement of Biogeochemical Rates

Stream metabolism—During each field campaign, ER and GPP were calculated in each reach from at least 3 days of continuous DO concentration data using the two-station open channel method (Young and Huryn 1998). DO concentrations and stream temperature were measured at 5 min intervals using deployable sondes equipped with optical DO probes (Yellow Springs Instruments 600OMS-O) at the upstream and downstream ends of each reach. Before deployment, DO probes were calibrated in the field following manufacturer's instructions. Following calibration, the two probes were kept together in the stream for 1 h to confirm consistency of readings. Photosynthetically active radiation sensors (Odyssey, Inc.) were deployed concurrently with the sondes to identify daytime and nighttime measurements.

To calculate metabolic parameters from DO fluxes, atmospheric exchange was estimated using the nighttime regression method of Kelly et al. (1974). Changes in DO concentrations at night result only from ER and reaeration, so the first derivative of DO with respect to time can be expressed as:

$$\frac{dC}{dt} = -R + k_{O_2}(C_s - C) \quad (1)$$

where $\frac{dC}{dt}$ is the change in DO over time, R is ecosystem respiration, k_{O_2} is the reaeration coefficient, C is the in situ DO concentration, C_s is the DO concentration at saturation, and $(C_s - C)$ is the DO deficit. A plot of $\frac{dC}{dt}$ against $(C_s - C)$ will yield a positive slope of k_{O_2} .

After k_{O_2} was determined, the reaeration DO flux was calculated from k_{O_2} , the DO deficit, and temperature. By subtracting the reaeration flux from the observed DO flux, the metabolic flux was calculated. ER was calculated from the average nighttime metabolic flux, which was adjusted to account for the effects of temperature on reaeration (Elmore and West 1961). GPP was then calculated as the difference between ER and metabolic flux during the day. Net ecosystem production (NEP) was calculated as the difference between ER and GPP for each 24-h period; negative values of NEP indicate more organic matter being respired

than produced. Metabolic rates were transformed from units of O_2 to units of carbon using the respiratory quotient of 0.85 mol CO_2 /mol O_2 (Odum 1956). Each daily measurement of metabolism was treated as an independent replicate for purposes of statistical analysis.

Nutrient retention

Whole-stream nutrient retention was measured using a pulse release method modified from Tank et al. (2008). Depending on the nutrient of interest, either KNO_3 , NH_4Cl , or KH_2PO_4 was mixed with ~100 L of stream water and 10–20 kg of NaCl, which served as a conservative tracer. Calculations based on previous measurements of dispersion were used to ensure that background nutrient concentrations were only enriched by 40–60 $\mu g/L$, as studies have shown that nutrient releases can underestimate nutrient retention if nutrient demand is saturated (Mulholland et al. 2002, Payn et al. 2005). Injectate was sampled and then released into the stream water as a single slug, but evenly across the stream width to promote full and rapid mixing. At a downstream station at a distance corresponding to a 20-min travel time, stream water was sampled at intervals and specific conductivity and time were recorded. Stream water was sampled at 0.5–1 min intervals as conductivity peaked and began to fall, then at 3–5 min intervals through the rest of the curve. Background-adjusted nutrient concentrations and specific conductivity were then plotted as a function of time and the area under these curves was determined (Fig. 3). The ratio of nutrient to conductivity in the injectate was compared to the ratio of the areas under the curves and used to calculate nutrient uptake lengths according to the equation:

$$S_w = \frac{L}{\ln\left(\frac{R_D}{R_I}\right)} \quad (2)$$

where S_w is nutrient uptake length, or the average distance a nutrient molecule travels before being removed from the water column. L represents the stream length in meters, R_D is the nutrient to conductivity ratio from the downstream sampling station, and R_I is the same ratio from the injectate. Nutrient uptake lengths are heavily influenced by discharge, so calculation of the uptake velocity (V_f) is used to standardize measurements of nutrient retention across different discharge conditions:

$$V_f = \frac{\bar{u} \times \bar{d}}{S_w} \quad (3)$$

where $\bar{u}$ and $\bar{d}$ represent the average water velocity and depth, respectively. The uptake velocity is analogous to a deposition velocity or piston velocity, and can be multiplied by the background nutrient concentration to estimate areal uptake rate (U), of nutrients in mass of nutrient per area of stream bed per time. Unlike the daily stream metabolism measurements, only one nutrient release was performed during each field campaign. Because of this, uptake metrics are point measurements.

Denitrification enzyme activity—Both potential and ambient denitrification enzyme activity (DEA) were measured in sediments in November. Ambient DEA is expected to reflect ambient denitrification rates, including N or C limitation, whereas potential DEA

reflects maximum denitrification rates that would be expected when nitrate and labile DOC (glucose) are added to the assays. Ambient and potential DEA were measured using the acetylene block technique, in which acetylene is used to inhibit the reduction of N_2O to N_2 , and rates are calculated from the accumulation of N_2O over time (e.g., Royer et al. 2004). Sediments were incubated with stream water collected at the same time and location as stream sediments. Following the assay, sediments were dried, combusted, and weighed as above to calculate DEA on a per gram AFDM basis.

Statistical Analysis

The solute release method used to estimate nutrient uptake provided a single reach-integrated measurement for each release, which precluded statistical analysis. For GPP and ER, consecutive days during each sampling period were used as replicates. We used independent sample *t* tests to compare the restored and unrestored reaches in ambient and potential DEA, because DEA was measured on only one date. The other variables were compared using two-way ANOVA with reach (restored vs. unrestored) and date as factors. The reach $\times$ date interaction term was also included in the model, and Tukey's post hoc test was used for pairwise comparisons between restored and unrestored reaches. Spearman correlation was used to test for significant monotonic relationships between variables. In all cases, differences were considered statistically significant when $p < 0.05$. All data were tested for normality with the Kolmogorov–Smirnov test. Variables that were not normally distributed were $\log_{10}$ transformed, which successfully normalized those variables. All statistical analyses were performed in Minitab version 17 (Minitab, Inc.).

Results

Sediment Properties

Sediment carbon content—Percent organic carbon increased and became more variable in both reaches throughout the study period, ranging from median values of 0.8–2.6% organic carbon in the restored reach and 0.9–1.9% organic carbon in the unrestored reach (Fig. 4a). Two-way ANOVA indicated significant effect of date ($F = 13.68$, $df = 3$, $p < 0.001$) but not of reach. There was weak evidence for a significant interaction ($p = 0.058$) and organic matter content was only different between reaches in November (Tukey's post hoc test, $p < 0.05$).

Phosphate equilibria—Mean EPC_0 ranged from 13.6–43.6 $\mu\text{g/L}$ in the restored reach, and from 5.5–18.5 $\mu\text{g/L}$ in the unrestored reach (Fig. 4b). Both reaches exhibited the same temporal trends, remaining low through June and July before a large rise in September, followed by a slight recession in November. Two-way ANOVA indicated significant main effects for date ($F = 7.33$, $df = 3$, $p < 0.001$) and reach ($F = 15.89$, $df = 3$, $p < 0.001$) but no interaction ($p = 0.939$). Mean EPC_0 values were always higher than background SRP concentrations in the restored reach, suggesting that sediments were likely acting as a source of P to the water column. Mean EPC_0 values were lower than or very close to background SRP concentrations in the unrestored reach for all samplings but September, suggesting that sediments were a potential sink for P from the water column.

Sediment oxygen demand—Mean SOD in the restored reach was 0.223 mg O₂/g AFDM/h in September and 0.015 mg O₂/g/h in November (Fig. 5). In the unrestored reach, mean oxygen demand was 0.509 mg O₂/g/h and 0.130 mg O₂/g/h. Both date and reach were significant main effects ($F = 61.3$, $df = 1$, $p < 0.001$ and $F = 26.96$, $df = 1$, $p < 0.001$, respectively) and there was a significant interaction term ($p = 0.048$). Tukey's post hoc test indicated SOD was significantly higher in the unrestored reach in September ($p < 0.05$) but not in November. SOD was significantly and inversely related to sediment percentage C (Table 2).

Biogeochemical rates

Stream metabolism—GPP in the unrestored reach peaked at >3 g C/m²/d during July before declining to <0.2 g C/m²/d in November (Fig. 6). Two-way ANOVA showed both date ($F = 80.01$, $df = 3$, $p < 0.001$) and reach ($F = 130.9$, $df = 1$, $p < 0.001$) were significant factors, and the interaction term was also significant ($p < 0.001$). GPP in the unrestored reach was greater than the restored reach in June and July (Tukey's post hoc test, $p < 0.05$), but not in September or November.

For ER, both date and reach were significant factors (two-way ANOVA with $F = 21.77$, $df = 3$, $p < 0.001$ and $F = 92.62$, $df = 1$, $p < 0.001$, respectively) and there was a significant interaction term ($p < 0.001$). ER was greater in the unrestored reach than the restored reach in June, July, and November (Tukey's post hoc test, $p < 0.05$), but not in September. On all dates both reaches had negative NEP (Table 3), indicating respiration exceeded primary production in Fawn River.

Nutrient retention—Total N concentrations ranged from 419–1322 µg/L, with nitrate accounting for 39–74% of total N. Ammonium S_w ranged from 484–772 m in July and September (Table 4). No retention could be measured in November, and ammonium S_w exceeded 3000 m in both reaches in June due to high water velocity and low uptake. Measured ammonium uptake velocities were lowest (0.20–2.74 mm/min or unmeasurable) in June and November and highest (5.97–11.51 mm/min) in July and September. Significant negative relationships were found between ammonium V_f and background ammonium concentration ($r_s = -0.866$, $n = 6$, $p = 0.0333$), nitrate concentration ($r_s = -0.943$, $n = 6$, $p = 0.0167$) and total N ($r_s = -0.943$, $n = 6$, $p = 0.0167$). Although the sample size is limited, these results indicate that ammonium V_f was affected by background N concentrations. Ammonium V_f and S_w were similar between the restored and unrestored reaches in July and September, but ammonium V_f was 13.7 times higher in the unrestored reach during June. Both the lowest and highest measured values of U (0.016 and 0.204 g N/m²/d) occurred during June in the restored and unrestored reaches, respectively. Out of four releases, nitrate retention was only observed in the unrestored reach during September ($S_w = 954$ m, $V_f = 3.17$ mm/min).

SRP concentrations varied little throughout the study period, ranging from 5.3 to 9.4 µg/L. TP concentrations ranged from 6.9 to 17.1 µg/L, decreasing throughout the study period. No obvious between-reach differences were found for SRP or TP concentrations. SRP S_w

ranged from 326 to 975 m and was higher in the unrestored reach in June, similar in July and November, and substantially lower in September (Table 5).

Denitrification enzyme activity—Mean ambient DEA was 15.5 $\mu\text{mol N/g AFDM/h}$ and 47.4 $\mu\text{mol N/g/h}$ in sediments from the restored and unrestored reach, respectively (Fig. 7a). Mean potential DEA was 24.5 and 99.0 $\mu\text{mol N/g/h}$ in restored and unrestored sediments (Fig. 7b). The unrestored reach had significantly higher ambient DEA ($t = -6.97$, $df = 16$, $p < 0.001$) and potential DEA ($t = -3.71$, $df = 16$, $p = 0.002$). Both measures of DEA were significantly and inversely related to sediment % C (Table 2).

Discussion

Fine sediment removal altered the magnitude and pattern of biogeochemical functions in Fawn River. Lower GPP in the restored reach during summer months indicates that fine sediment removal reduced GPP through the elimination of rooted macrophytes from the streambed. Fine sediment removal also reduced ER, suggesting that this may also be attributed in part to elimination of macrophytes. Effects of fine sediment removal on nutrient cycling seem to be more nuanced. Fine sediment removal appeared to have limited near-term effects on N retention at the reach scale, but DEA was reduced. Low values of EPC_0 combined with very high values of V_f for P uptake suggest that removal of fine sediment changed P retention in Fawn River by reducing sediments that support P adsorption.

Stream Metabolism

Submerged macrophytes colonized fine silt in Fawn River during the years following the reservoir sediment release (Lewis et al. 2013), and our results indicate that macrophyte presence is a primary control on GPP and ER. Very few submerged macrophytes were observed in the restored reach, likely due to dependence on fine sediment deposits for habitat. Our results suggest that fine sediment removal lowered GPP in Fawn River by removing macrophytes associated with fine sediment deposits and preventing their recolonization. Macrophytes directly increase GPP in freshwater ecosystems (Allen 1971, Likens 1975, Alnoe et al. 2016) and they can indirectly increase primary productivity by providing an attachment surface for epiphytic algae (Hooper and Robinson 1976, Cattaneo and Kalff 1980). ER was lower in the restored reach on all sampling dates but September, suggesting that fine sediment removal also reduced ER by removing macrophytes. Previous work has demonstrated that removal of macrophytes can decrease ER by reducing root respiration (e.g., Kaenel et al. 2000). Submerged macrophytes may also stimulate heterotrophic respiration in their rooting zone through the release of oxygen (Sand-Jensen et al. 1982) or organic-rich exudates (Karjalainen et al. 2001). By influencing the magnitude and seasonal patterns of metabolic processes in Fawn River, fine sediment removal could influence the types and amounts of food resources available for consumers. Removing rooted macrophytes and fine sediments should result in a shift in the base of the food web from vascular macrophytes to periphyton, and presumably also a shift in the types of macroinvertebrates present in the benthic community. Shifts in primary production from vascular macrophytes to presumably periphyton may influence higher trophic levels by

shifting the composition of functional feeding groups and habitat associations to a more diverse community able to occupy a wider range of trophic and habitat niches.

Removing macrophytes from Fawn River appears to be a driving factor behind observed differences in metabolism, but other factors could have also contributed. Outside of the effects of fine sediment removal, natural geomorphic differences between the restored and unrestored reach (i.e. those not resulting from fine sediment removal) may have influenced stream metabolism. The unrestored reach was slower, deeper, and wider than the restored reach (Table 1), which may have resulted in naturally higher stream respiration due to longer water residence times and a larger benthic surface area (Mulholland et al. 2001). Differences in sediment organic matter bioavailability may have contributed to higher ER in the unrestored reach. SOD was higher in the unrestored reach for both the September and November samplings (Fig. 5), suggesting that organic matter bioavailability may have been reduced by fine sediment removal, although data was not available for the June and July samplings. Fine sediment removal may also alter ER through changing the composition of bacterial and faunal communities. Different species and functional traits have higher rates of respiration or are active during different seasons, which could contribute to the differences observed. However, we did not collect species composition data on bacterial or faunal communities in Fawn River and cannot assess whether this may explain observed patterns. Lastly, fine sediment removal may increase heterotrophic respiration by deepening the hyporheic zone. In a companion paper, Ward et al. (2018) found that fine sediment removal at Fawn River may have increased hyporheic exchange during high flows. ER was lower in the restored reach during all but the low flow period in September (Fig. 6), suggesting that any stimulating effect of fine sediment removal on hyporheic metabolism was minimal and/or offset by other mechanisms in the short term. This may be related to low discharge in Fawn River during this time, when the restored reach became a gaining system (Table 1), and a larger proportion of the flow was subsurface (Ward et al. 2018).

Nutrient Retention

Watershed and sediment controls on N retention—General patterns in N retention did not demonstrate obvious effects of restoration at the reach scale. The lack of observed nitrate retention, as well as negative correlations of ammonium V_f with ammonium, nitrate and total N concentrations, indicate that N retention is strongly influenced by nitrogen availability in the water column, which is under watershed-scale control and therefore unlikely to be affected by reach-scale restoration (Bernhardt and Palmer 2011). A large proportion of the Fawn River watershed area is used for row-crop agriculture, with seasonal applications of N fertilizers. The wide fluctuations in N availability observed throughout the study (Table 4) may be caused by these watershed characteristics (e.g., Carpenter et al. 1998), obscuring any small between-reach differences in N retention. This highlights the importance of spatial context when selecting sites for restoration and the limitations of channel-oriented restoration practices (Bernhardt and Palmer 2011)—while channel restoration may increase or decrease potential for N retention, watershed-level controls on N availability likely drive N dynamics in Fawn River by saturating N retention processes when fertilizer runoff is high.

Despite the lack of observations of nitrate retention, DEA was measurable in all sediment samples in November, the only field campaign where it was assessed. Because DEA was only assessed in November, few conclusions can be drawn about the effects of fine sediment removal on denitrification in streambed sediments. Both ambient and potential DEA were higher in the unrestored reach, suggesting that fine sediment removal may reduce denitrification by lowering the activity of denitrifying bacterial communities. Observed relationships between DEA and other sediment properties suggest that these differences may be mediated by macrophyte presence and/or effects on sediment organic matter. Forshay and Dodson (2011) found high DEA in sediments associated with macrophyte roots, and that DEA was not carbon-limited. Arango et al. (2007) found that quality of sediment organic matter can influence denitrification rates, particularly when nitrate concentrations are high, and it appears this may have been a factor influencing DEA in Fawn River. Ambient DEA was positively correlated with SOD (Table 2), indicating that denitrification was facilitated by similar sediment properties as aerobic respiration. Our results also suggest that high DEA in sediments associated with macrophyte roots may be mediated by macrophyte effects on sediment organic carbon availability.

Despite higher values of DEA in the unrestored reach, no nitrate retention was observed during this study, suggesting that a compensating mechanism may be introducing nitrate to the water column. High rates of sediment respiration in the unrestored reach may create anoxic conditions, stimulating the release of ammonium adsorbed to sediments (e.g., Beutel 2006), where it can be transformed into nitrate via nitrification. Nitrate concentrations were higher in the unrestored reach during all samplings (Table 4), suggesting that this mechanism may be a significant source of N to the water column. Even as fine sediment removal may have reduced DEA in the restored reach, it likely would reduce ammonium release from anoxic sediments by physically removing the source of ammonium and alleviating anoxic conditions in the hyporheic zone.

Sediment controls on P retention—P retention rates measured in both restored and unrestored reaches of Fawn River were as much as two orders of magnitude higher than those reported in studies using steady-state releases of P (e.g., Ensign and Doyle 2006), and 2–4 times higher than values reported in other pulse addition experiments (Powers et al. 2009, Griffiths and Johnson 2018). Powers et al. (2009) compared P retention metrics from steady-state and pulse releases and found that metrics calculated using pulse releases were consistently higher than metrics from steady-state releases, attributing the difference to P adsorption to sediments. Our results suggest that high adsorption of P by streambed sediments, particularly in the unrestored reach, controlled P retention in Fawn River. We believe adsorption could be responsible for most phosphorus retention measured in this study. A recent review of legacy phosphorus emphasized that P adsorption and desorption in fine sediments can be the dominant control on stream P concentrations (Sharpley et al. 2013). Median EPC_0 was lower in the unrestored reach on all sampling dates and usually at or below the measured stream water SRP concentration (Fig. 4b, Table 5), suggesting that stream water P concentrations were strongly affected by adsorption to sediments. By increasing EPC_0 , fine sediment removal likely reduced the capacity of benthic sediments to adsorb P from the water column.

Due to the apparent abiotic control of P retention in Fawn River, little can be inferred about differences in biotic demand for P between the restored and unrestored reaches. Whole-stream nutrient releases to measure retention rates operate under the assumption that adsorption and desorption are at equilibrium and measured nutrient retention is exclusively biological. However, our results suggest that this assumption may not be true in rivers with large amounts of fine sediment, where adsorption of phosphate by sediments may be the dominant control on stream water P concentrations (e.g., Lottig and Stanley 2007). It is surprising that SRP V_f was high in both reaches relative to literature values, even though the restored reach had most fine sediments removed. The restored reach may have had higher levels of biotic P retention processes which compensated for the loss of P adsorption to fine sediments. Conversely, the fine sediments not removed from the stream during restoration (e.g., left in place in natural deposition areas or along channel margins) may have played a large role in reach-scale P retention.

Management Implications

This study has implications for predicting the biogeochemical effects of fine sediment and macrophyte removal, as well as the utility of reach-scale stream restoration as a tool to manage biogeochemical functions at broader scales. Fine sediment and macrophyte removal are typically management actions geared towards improving fish habitat, hydrology, and aesthetics rather than biogeochemical functions. However, this study suggests that these practices can also result in alterations to the magnitude and seasonal patterns of stream metabolism and may also affect nutrient retention. We found limited evidence for a direct effect of fine sediment removal on stream metabolism, instead finding that the removal of macrophytes was the primary influence on GPP and ecosystem respiration in the Fawn River restoration project. This suggests that measures to remove macrophytes or control their growth can be immediately effective at restoring natural patterns of GPP and ER to sediment-impaired streams. However, fine sediment deposits are highly susceptible to macrophyte recolonization, so destabilizing and removing these deposits will be necessary to ensure that biogeochemical changes are persistent.

Fine sediment removal appears to have reduced the magnitude of denitrification and P adsorption in sediments of Fawn River; this suggests that some restoration practices may lead to reductions in desirable biogeochemical processes. However, the limited response of reach-scale N and P retention to sediment removal suggests that compensating mechanisms may offset these reductions, at least over the temporal scale of this study. To fully account for the variety of processes governing emergent biogeochemical functions such as ecosystem metabolism and nutrient cycling, we recommend that a full suite of specific biogeochemical processes of interest (e.g. denitrification, N adsorption and release from sediments, and assimilatory N uptake), as well as measurements of ecosystem structure including macrophyte cover and periphyton biomass, should be assessed when planning restoration projects.

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References

- Allen HL (1971) Primary productivity, chemo-organotrophy, and nutritional interactions of epiphytic algae and bacteria on macrophytes in the littoral of a lake. *Ecol Monogr* 41(2):97–127. 10.2307/1942387
- Alnoe AB, Riis T, Baattrup-Pedersen A (2016) Comparison of metabolic rates among macrophyte and nonmacrophyte habitats in streams. *Freshw Sci* 35:834–844. 10.1086/687842
- Anderson DM, Gilbert PM, Burkholder JM (2002) Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries* 25(4):704–726. 10.1007/BF02804901
- Arango CP, Tank JL, Schaller JS, Royer TV, Bernot MJ, David MB (2007) Benthic organic carbon influences denitrification in streams with high nitrate concentration. *Freshw Biol* 52:1210–1222. 10.1111/j.1365-2427.2007.01758.x
- Balestrini R, Delconte CA, Palumbo MT, Buffagni A (2018) Biotic control of in-stream nutrient retention in nitrogen-rich springs (Po Valley, Northern Italy). *Ecol Eng* 122:303–314. 10.1016/j.ecoleng.2018.08.014
- Beckett PHT, White RE (1964) Studies on the phosphate potentials of soils. *Plant Soil* 21(3):253–282. 10.1007/BF01377744
- Bernhardt ES, Palmer MA, Allan JD, Alexander G, Barnas K, Brooks S, Carr J, Clayton S, Dahm C, Follstad-Shah J, Galat D (2005) Synthesizing US river restoration efforts. *Science* 308 (5722):636–637. 10.1126/science.1109769 [PubMed: 15860611]
- Bernhardt ES, Palmer MA (2011) River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. *Ecol Appl* 21(6):1926–1931. 10.1890/10-1574.1 [PubMed: 21939034]
- Beutel MW (2006) Inhibition of ammonia release from anoxic profundal sediments in lakes using hypolimnetic oxygenation. *Ecol Eng* 28(3):271–279. 10.1016/j.ecoleng.2006.05.009
- Bukaveckas PA (2007) Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. *Env Sci Technol* 41(5):1570–1576. 10.1021/es061618x [PubMed: 17396643]
- Cabrera ML, Beare MH (1993) Alkaline persulfate oxidation for determining total nitrogen in microbial biomass extracts. *Soil Sci Soc Am J* 57(4):1007–1012. 10.2136/sssaj1993.03615995005700040021x
- Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol Appl* 8(3):559–568. 10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2
- Cattaneo A, Kalff J (1980) Relative contribution of aquatic macrophytes and their epiphytes to the production of macrophyte beds. *Limnol Oceano* 25(2):280–289. 10.4319/lo.1980.25.2.0280
- Collins AL, Walling DE (2007) Sources of fine sediment recovered from the channel bed of lowland groundwater-fed catchments in the UK. *Geomorphol* 88(1–2):120–138. 10.1016/j.geomorph.2006.10.018
- Craig LS, Palmer MA, Richardson DC, Filoso S, Bernhardt ES, Bledsoe BP, Doyle MW, Groffman PM, Hassett BA, Kaushal SS, Mayer PM, Smith SM, Wilcock PR (2008) Stream restoration strategies for reducing river nitrogen loads. *Front Ecol Environ* 6 (10):529–538. 10.1890/070080
- Dodds WK, Biggs BJ (2002) Water velocity attenuation by stream periphyton and macrophytes in relation to growth form and architecture. *J N Am Benthol Soc* 21(1):2–15. 10.2307/1468295
- Downes BJ, Barmuta LA, Fairweather PG, Faith DP, Keough MJ, Lake PS, Mapstone BD, Quinn GP (2002) Monitoring ecological impacts: Concepts and practice in flowing waters. Cambridge University Press, UK. 10.1017/CBO9780511542015
- Elmore HL, West WF (1961) Effects of water temperature on stream reaeration. *J Sanit Eng Div Am Soc Civ Eng* 87:SA–6
- Ensign SH, Doyle MW (2006) Nutrient spiraling in streams and river networks. *J Geophys Res Biogeosciences* 111(G4). 10.1029/2005JG000114

- Fellows CS, Valett HM, Dahm CN (2001) Whole-stream metabolism in two montane streams: contribution of the hyporheic zone. *Limnol Oceanogr* 46:523–531. 10.4319/lo.2001.46.3.0523
- Forshay KJ, Dodson SI (2011) Macrophyte presence is an indicator of enhanced denitrification and nitrification in sediments of a temperate restored agricultural stream. *Hydrobiol* 668(1):21–34. 10.1007/s10750-011-0619-2
- Gales ME, Julian EC, Kroner RC (1966) Method for quantitative determination of total phosphorus in water. *J Am Water Works Assoc* 58(10):1363–1368. 10.1002/j.1551-8833.1966.tb01703.x
- Griffiths NA, Johnson LT (2018) Influence of dual nitrogen and phosphorus additions on nutrient uptake and saturation kinetics in a forested headwater stream. *Freshw Sci* 37(4):810–825. 10.1086/700700
- Gücker B, Boëchat IG (2004) Stream morphology controls ammonium retention in tropical headwaters. *Ecology* 85(10):2818–2827. 10.1890/04-0171
- Hall RO (2016) Metabolism of streams and rivers: estimation, controls, and applications In: Jones JB, Stanley EH (eds) *Stream ecosystems in a changing environment*, 1st edn, Elsevier, New York, pp 151–180
- Hill BH, Hall RK, Husby P, Herlihy AT, Dunne M (2000) Interregional comparisons of sediment microbial respiration in streams. *Freshw Biol* 44(2):213–222. 10.1046/j.1365-2427.2000.00555.x
- Hooper NM, Robinson GGC (1976) Primary production of epiphytic algae in a marsh pond. *Can J Bot* 54(24):2810–2815. 10.1139/b76-300
- Hurlbert SH (2004) On misinterpretations of pseudoreplication and related matters: a reply to Oksanen. *Oikos* 104:591–597. 10.1111/j.0030-1299.2004.12752.x
- Kaenel BR, Buehrer H, Uehlinger U (2000) Effects of aquatic plant management on stream metabolism and oxygen balance in streams. *Freshw Biol* 45(1):85–95. 10.1046/j.1365-2427.2000.00618.x
- Karjalainen H, Stefansdottir G, Tuominen L, Kairesalo T (2001) Do submersed plants enhance microbial activity in sediment? *Aquat Bot* 69(1):1–13. 10.1016/S0304-3770(00)00138-8
- Kelly MG, Hornberger GM, Cosby BI (1974) Continuous automated measurement of rates of photosynthesis and respiration in an undisturbed river community. *Limnol Oceanogr* 19(2):305–312. 10.4319/lo.1974.19.2.0305
- Lewis N, Priore B, Lewis C, Nicholas D, Tucker-Hahn E (2013) Fawn river restoration. *Land Water May/June* 2013:49–52
- Likens GE (1975) Primary production of inland aquatic ecosystems In: Lieth H, Whittaker RH (eds) *Primary productivity of the biosphere*, 1st edn, Springer, Heidelberg, p 185–202
- Lindsey AA, Schmalz DV, Nichols SA (1970) Natural areas in Indiana and their preservation. Indiana Natural Areas Survey, Dept. of Biological Sciences, Purdue University, West Lafayette
- Lottig NR, Stanley EH (2007) Benthic sediment influence on dissolved phosphorus concentrations in a headwater stream. *Biogeochemistry* 84(3):297–309. 10.1007/s10533-007-9116-0
- Mendoza-Lera C, Detry T (2017) Relating hydraulic conductivity and hyporheic zone biogeochemical processing to conserve and restore river ecosystem services. *Sci Tot Environ* 579:1815–1821. 10.1016/j.scitotenv.2016.11.166
- Mulholland PJ, Fellows CS, Tank JL, Grimm NB, Webster JR, Hamilton SK, Marti E, Ashkenas L, Bowden WB, Dodds WK, McDowell WH (2001) Inter-biome comparison of factors controlling stream metabolism. *Freshw Biol* 46(11):1503–1517. 10.1046/j.1365-2427.2001.00773.x
- Mulholland PJ, Tank JL, Webster JR, Bowden WB, Dodds WK, Gregory SV, Grimm NB, Hamilton SK, Johnson SL, Marti E, McDowell WH, Merriam JL, Meyer JL, Peterson BJ, Valett HM, Wollheim WM (2002) Can uptake length in streams be determined by nutrient addition experiments? Results from an interbiome comparison study. *J N Am Benthol Soc* 21(4):544–560. 10.2307/1468429
- Mulholland PJ, Helton AM, Poole GC, Hall RO, Hamilton SK, Peterson BJ, Tank JL, Ashkenas LR, Cooper LW, Dahm CN, Dodds WK, Findlay SEG, Gregory SV, Grimm NB, Johnson SL, McDowell WH, Meyer JL, Valett HM, Webster JR, Arango CP, Beaulieu JJ, Bernot MJ, Burgin AJ, Crenshaw CL, Johnson LT, Niederlehner BR, O'Brien JM, Potter JD, Sheibley RW, Sobota DJ, Thomas SM (2008) Stream denitrification across biomes and its response to anthropogenic nitrate loading. *Nature* 452:202–206. 10.1038/nature06686 [PubMed: 18337819]

- Newcomer-Johnson TA, Kaushal SS, Mayer PM, Smith R, Sivirichi G (2016) Nutrient retention in restored streams and rivers: A global review and synthesis. *Water* 8(4):116 10.3390/w8040116
- Newcomer TA, Kaushal SS, Mayer PM, Shields AR, Canuel EA, Groffman PM, Gold AJ (2012) Influence of natural and novel organic carbon sources on denitrification in forest, degraded urban, and restored streams. *Ecol Monogr* 82:449–466. 10.1890/12-0458.1
- Odum HT (1956) Primary production in flowing waters. *Limnol Oceano* 1(2):102–117. 10.4319/lo.1956.1.2.0102
- Paerl HW, Pinckney JL, Fear JM, Peierls BL (1998) Ecosystem responses to internal and watershed organic matter loading: consequences for hypoxia in the eutrophying Neuse River Estuary, North Carolina, USA. *Mar Ecol Prog Ser* 166:17–25
- Palmer MA, Hondula KL, Koch BJ (2014) Ecological restoration of streams and rivers: shifting strategies and shifting goals. *Annu Rev Ecol Syst* 45:247–269. 10.1146/annurev-ecolsys-120213-091935
- Palmer MA, Bernhardt ES, Allan JD, Lake PS, Alexander G, Brooks S, Carr J, Clayton S, Dahm CN, Follstad-Shah J, Galat DL (2005) Standards for ecologically successful river restoration. *J Appl Ecol* 42(2):208–217. 10.1111/j.1365-2664.2005.01004.x
- Payn RA, Webster JR, Mulholland PJ, Valett H, Dodds WK (2005) Estimation of stream nutrient uptake from nutrient addition experiments. *Limnol Oceano Methods* 3:174–182. 10.4319/lom.2005.3.174
- Powers SM, Stanley EH, Lottig NR (2009) Quantifying phosphorus uptake using pulse and steady-state approaches in streams. *Limnol Oceano Methods* 7(7):498–508. 10.4319/lom.2009.7.498
- Royer TV, Tank JL, David MB (2004) Transport and fate of nitrate in headwater agricultural streams in Illinois. *J Env Qual* 33 (4):1296–1304. 10.2134/jeq2004.1296 [PubMed: 15254111]
- Sand-Jensen K, Prahl C, Stokholm H (1982) Oxygen release from roots of submerged aquatic macrophytes. *Oikos* 38(3):349–354. 10.2307/3544675
- Schmadel NM, Ward AS, Kurz MJ, Fleckenstein JH, Zarnetske JP, Hannah DM, Blume T, Vieweg M, Blaen PJ, Schmidt C, Knapp JLA, Klaar MJ, Romeijn P, Datry T, Keller T, Folegot S, Marruedo Arricibita AI, Krause S (2016) Stream solute tracer timescales changing with discharge and reach length confound process interpretation. *Wat Res* 52:3227–3245. 10.1002/2015WR018062
- Sepulveda AJ, Sechrist J, Marczak LB (2014) Testing ecological trade-offs of a new tool for removing fine sediment in a spring-fed stream. *Ecol Restor* 32(1):68–77. 10.3368/er.32.1.68
- Sharpley A, Jarvie H, Buda A, May L, Spears B, Kleinman P (2013) Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. *J Environ Qual* 42:1308–1326. 10.2134/jeq2013.03.0098 [PubMed: 24216410]
- Tank JL, Rosi-Marshall EJ, Baker MA, Hall RO (2008) Are rivers just big streams? A pulse method to quantify nitrogen demand in a large river. *Ecology* 89(10):2935–2945. 10.1890/07-1315.1 [PubMed: 18959330]
- Valett HM, Crenshaw CL, Wagner PF (2002) Stream nutrient uptake, forest succession, and biogeochemical theory. *Ecology* 83:2888–2901. 10.1890/0012-9658(2002)083[2888:SNUFSA]2.0.CO;2
- Ward AS, Morgan JA, White JR, Royer TV (2018) Streambed restoration to remove fine sediment alters reach-scale transient storage in a low-gradient 5th order river, Indiana, USA. *Hydrol Proc* 32(12):1786–1800. 10.1002/hyp.11518
- Waters TF (1995) Sediment in streams: sources, biological effects, and control. Vol 7 American Fisheries Society, Bethesda
- Wesley JK (2000) Fawn River (Middle) surveyed July 27 and 28, 1994. Michigan Department of Natural Resources Status of the Fishery Resource Report 2000–2007.
- Young RG, Huryn AD (1998) Comment: improvements to the diurnal upstream-downstream dissolved oxygen change technique for determining whole-stream metabolism in small streams. *Can J Fish Aquat Sci* 55(7):1784 10.1139/f98-052

Fig. 1.
a Before and after photos of the same location on the streambed showing the removal of fine sediment. Images are ~40 cm in width. **b** Map of the study area indicating the locations of the restored and unrestored reaches of Fawn River. Inset map of Indiana (USA) shows Steuben County, the location of the study site, in black. Map is redrawn from Ward et al. (2018). Images in a provided by Fawn River Restoration and Conservation Charitable Trust

Fig. 2.

Example isotherm curves from November 2013 in the **a** restored and **b** unrestored reaches showing how the equilibrium phosphate concentration for streambed sediments (EPC_0) was estimated using quadratic regression. EPC_0 is the stream water concentration at which there is no net exchange with the sediments and solving for x when $y = 0$; this is shown above as the intersection of the dashed lines

Fig. 3.

Example breakthrough curve for soluble reactive phosphorus (SRP) and conductivity (as a proxy for the conservative tracer, chloride) following a pulse release in July 2013 in Fawn River. The uptake of SRP can be determined by integrating the area under the curve as the pulse moves downstream (see text for details)

Fig. 4.

a Sediment % C and **(b)** EPC₀ for sediments from restored and unrestored reaches of Fawn River on four dates in 2013. Data are presented as box and whisker plots in which the center horizontal line is the median, the box represents the 25th and 75th percentiles, the whiskers represent the 10th and 90th percentiles; the horizontal dashed line is the mean. If the mean line is not visible it is equal to the median, except for restored EPC₀ in September when the mean was 44 μg SRP L⁻¹. An asterisk indicates a significant difference ($p < 0.05$) between restored and unrestored reaches within a date based on Tukey's post hoc test

Fig. 5. Sediment oxygen demand from restored and unrestored reaches of Fawn River on two dates in 2013. Box and whisker plots are as described in Fig. 4. An asterisk indicates a significant difference ($p < 0.05$) between restored and unrestored reaches within a date based on Tukey's post hoc test

Fig. 6. Mean (± 1 SE) gross primary productivity and ecosystem respiration in the restored and unrestored reaches of Fawn River during four dates in 2013. An asterisk indicates a significant difference ($p < 0.05$) between restored and unrestored reaches within a date based on Tukey's post-hoc test

Fig. 7. **a** Ambient and **b** potential denitrification enzyme activity (DEA) in restored and unrestored sediments from Fawn River during November 2013. Box and whisker plots are as described in Fig. 4. *p*-values are from independent sample *t* tests between restored and unrestored reaches

Table 1

Mean discharge (Q), upstream-downstream change in discharge (Q), reach length (L), width (w), depth (d), and velocity (V) in restored and unrestored reaches of Fawn River on four dates in 2013

Date	Reach	Q (m ³ /s)	Q (m ³ /s, %)	L (m)	w (m)	d (m)	V (m/s)
June	Restored	2.46	-0.20, -9%	407	10.4	0.57	0.41
	Unrestored	2.33	-0.50, -26%	333	12.3	0.72	0.26
July	Restored	1.40	0.04, 3%	347	11.1	0.46	0.27
	Unrestored	1.80	-0.26, -14%	297	12.5	0.86	0.17
Sept	Restored	0.32	0.05, 15%	222	10.2	0.30	0.10
	Unrestored	0.42	0.00, 0%	242	13.0	0.44	0.07
Nov	Restored	1.33	-0.10, -7%	345	10.1	0.42	0.31
	Unrestored	1.37	-0.26, -19%	314	15.6	0.75	0.12

Values are means of 3–5 days of consecutive monitoring

Table 2

Spearman correlation coefficients (r_s) for relationships between measures of sediment chemistry and biological processes in the sediment

	<i>N</i>	% <i>C</i>	SOD
SOD	36	-0.425 (0.010)	
Ambient DEA	18	-0.583 (0.011)	0.472 (0.047)
Potential DEA	18	-0.769 (<0.001)	0.370 (0.127)

Samples are pooled across the restored and unrestored reaches

N = sample size

p-values from the correlation analysis are in parentheses

Table 3

Mean gross primary productivity (GPP), ecosystem respiration (ER), and net ecosystem production (NEP) in restored and unrestored reaches of Fawn River on four dates in 2013

Date	Reach	GPP (g C/m ² /d)	ER (g C/m ² /d)	NEP (g C/m ² /d)
June	Restored	0.37	1.32	-0.96
	Unrestored	0.99	3.79	-2.80
July	Restored	0.59	1.86	-1.27
	Unrestored	3.20	3.94	-0.74
Sept	Restored	0.67	3.36	-2.69
	Unrestored	0.94	3.04	-2.11
Nov	Restored	0.13	3.02	-2.89
	Unrestored	0.14	5.85	-5.71

Values are means of 3–5 days of consecutive monitoring

Table 4

Ammonium, nitrate, and total N concentrations and ammonium uptake length (S_w), uptake velocity (V_f), and areal uptake rate (U) in restored and unrestored reaches of Fawn River on four dates in 2013

Month	Reach	$\text{NH}_4^+\text{-N}$ ($\mu\text{g/L}$)	$\text{NO}_3^-\text{-N}$ ($\mu\text{g/L}$)	Total N ($\mu\text{g/L}$)	$\text{NH}_4^+ S_w$ (m)	$\text{NH}_4^+ V_f$ (mm/min)	$\text{NH}_4^+ U$ (g N/m ² /day)
June	Restored	55	756	1016	>3000	0.20	0.016
	Unrestored	60	912	1322	>3000	2.74	0.204
July	Restored	6	185	419	670	11.51	0.083
	Unrestored	11	205	442	772	10.95	0.079
Sept	Restored	26	471	651	484	6.11	0.115
	Unrestored	23	580	860	506	5.97	0.106
Nov	Restored	98	359	920	No uptake observed		
	Unrestored	87	416	1011	No uptake observed		

Concentrations are mean values of 3–5 daily measurements in each reach. Spiraling metrics are single, reach-integrated values

Table 5

Soluble reactive phosphorus (SRP) and total P concentrations, and SRP uptake length (S_w), uptake velocity (V_f), and areal uptake rate (U) in restored and unrestored reaches of Fawn River on four dates in 2013

Month	Reach	SRP ($\mu\text{g/L}$)	TP ($\mu\text{g/L}$)	SRP S_w (m)	SRP V_f (mm/min)	SRP U (g/m ² /day)
June	Restored	6.3	17.1	821	13.1	0.115
	Unrestored	5.3	15.8	975	10.4	0.094
July	Restored	7.9	14.3	593	13.0	0.155
	Unrestored	8.6	12.5	588	14.4	0.193
Sept	Restored	9.4	10.0	564	5.2	0.075
	Unrestored	7.2	8.7	326	9.3	0.152
Nov	Restored	5.8	6.9	487	10.1	0.101
	Unrestored	6.8	8.6	482	20.2	0.175

Concentrations are mean values of 3–5 measurements in each reach. Spiraling metrics are single, reach-integrated values