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Seeing the light: urban stream restoration affects stream metabolism and nitrate uptake via changes in canopy cover

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Abstract. The continually increasing global population residing in urban landscapes impacts numerous ecosystem functions and services provided by urban streams. Urban stream restoration is often employed to offset these impacts and conserve or enhance the various functions and services these streams provide. Despite the assumption that "if you build it, [the function] will come," current understanding of the effects of urban stream restoration on stream ecosystem functions are based on short term studies that may not capture variation in restoration effectiveness over time. We quantified the impact of stream restoration on nutrient and energy dynamics of urban streams by studying 10 urban stream reaches (five restored, five unrestored) in the Baltimore, Maryland, USA, region over a two-year period. We measured gross primary production (GPP) and ecosystem respiration (ER) at the whole-stream scale continuously throughout the study and nitrate (NO_3^--N) spiraling rates seasonally (spring, summer, autumn) across all reaches. There was no significant restoration effect on NO_3^--N spiraling across reaches. However, there was a significant canopy cover effect on NO_3^--N spiraling, and directly comparing paired sets of unrestored-restored reaches showed that restoration does affect NO_3^--N spiraling after accounting for other environmental variation. Furthermore, there was a change in GPP : ER seasonality, with restored and open-canopied reaches exhibiting higher GPP : ER during summer. The restoration effect, though, appears contingent upon altered canopy cover, which is likely to be a temporary effect of restoration and is a driver of multiple ecosystem services, e.g., habitat, riparian nutrient processing. Our results suggest that decision-making about stream restoration, including evaluations of nutrient benefits, clearly needs to consider spatial and temporal dynamics of canopy cover and trade-offs among multiple ecosystem services.

Key words: canopy cover; ecosystem respiration; gross primary production; restoration; spiraling; urban stream.

Introduction

The degradation of the physical, chemical, and biological environment of stream ecosystems is a classic signature of the ecological impacts of urbanization (Walsh et al. 2005, Kaushal and Belt 2012). With the majority of the global population now living in urban areas (Grimm et al. 2008), there is great interest in understanding the impacts of urbanization on stream ecosystems and potential strategies to mitigate these impacts. The impacts of urbanization on streams, commonly referred to as the urban stream syndrome (Walsh et al. 2005), include geomorphic alterations such as highly incised banks and homogenized channel morphologies (Wolman 1967, Vietz et al. 2016). These geomorphological changes, coupled with increased nutrient loading caused by human activities in urban watersheds (Hatt et al. 2004, Carle et al. 2005, Bernhardt et al. 2008), lead to degradation of urban stream water quality and increased nutrient export. These ecological impacts of the urban stream syndrome, combined with the loss of other valuable ecological and social services provided by urban streams (Bolund and Hunhammer 1999), have led to major stream restoration efforts in urban areas, in an attempt to retain or enhance some of the functionality of these urban ecosystems (Bernhardt and Palmer 2007, Newcomer Johnson et al. 2016).

Despite multiple physical, chemical, and biological stressors (Walsh et al. 2005, Wenger et al. 2009), urban streams remain capable of providing high rates of various ecosystem functions, even in unrestored states

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(Reisinger et al. 2016). Nitrogen (N) cycling, in particular, represents a critical function of stream ecosystems across a range of human land uses (Mulholland et al. 2008). Understanding the capacity of urban streams to retain and remove nutrients prior to downstream export, and strategies to maximize these nutrient retention processes, is vital to reduce nutrient pollution of sensitive receiving water bodies. A review of N cycling across urban ecosystems found that various N spiraling metrics, which quantify different aspects of N cycling within streams (Stream Solute Workshop 1990), were as high or higher in urban streams compared to reference streams (Reisinger et al. 2016). This result shows that in spite of multiple stressors associated with the urban stream syndrome, urban streams remain capable of high rates of N uptake and removal. Additionally, a review of N uptake in restored streams found that nitrate uptake metrics were positively related to watershed impervious surface cover, and that stream restoration increased N spiraling activity relative to degraded urban streams (Newcomer Johnson et al. 2016).

Restoration of streams and rivers is a major economic investment, with billions of dollars spent each year in the United States to reduce the negative impacts of urbanization on stream ecosystems (Bernhardt et al. 2005, Bernhardt and Palmer 2007). Stream restoration practices often focus on channel stabilization and preventing bank erosion, assuming that stabilizing the geomorphology of the channel will minimize immediate threats of erosion to the built environment and infrastructure. Moreover, it is assumed that a stable stream channel will enhance stream ecological functionality (Gregory 2006, McMillan and Noe 2017). Despite the large economic investment, the effect of urban stream restoration on N removal, and the mechanisms driving this effect, remain unclear and warrant further exploration (Craig et al. 2008, Filoso and Palmer 2011, Newcomer Johnson et al. 2016).

Nitrogen removal within streams is primarily achieved by either biotic assimilation or denitrification, with denitrification representing a permanent removal of N from the ecosystem whereas biotic assimilation is a change to a less reactive N form and potentially delays downstream N export. Total N removal and denitrification rates are typically higher in anthropogenically impacted streams (agricultural and urban) than in reference sites (Mulholland et al. 2008, 2009, Hall et al. 2009), although removal efficiency (i.e., the proportion of total N load that is removed) decreases with increasing loads (Mulholland et al. 2008). Some forms of stream restoration aim to increase hydrologic connectivity and residence times through floodplain reconnection, allowing for increased contact time between microbial communities and stream water, increasing the potential for N to be removed via denitrification or other N cycling processes (Kaushal et al. 2008, Roley et al. 2012, McMillan and Noe 2017). Stream N removal is often controlled by stream metabolic activity (Hall and Tank 2003, Hall et al. 2009), which represents the production and consumption of energy by stream food webs (Bernhardt et al. 2018). These metabolic processes include the conversion of solar radiation into biomass and organic energy via gross primary production (GPP) and the consumption of this energy via both autotrophic and heterotrophic processes, collectively represented by ecosystem respiration (ER). The importance of stream metabolic activity and biotic assimilation driving N removal is particularly evident in urban streams (Arango et al. 2015, Ledford et al. 2017), which can exhibit high rates of GPP and ER despite frequent disturbance events due to high light and nutrient levels in these streams (Beaulieu et al. 2013, Reisinger et al. 2017).

Recent technological and analytical advances have allowed stream and river ecologists to overcome various logistical challenges inherent to estimating stream metabolism (Appling et al. 2018*a*, Bernhardt et al. 2018). These advances include the development of durable and economically affordable dissolved oxygen (DO) sensors, allowing researchers to deploy DO logging sensors across a range of field conditions for extended periods of time (e.g., Roberts et al. 2007, Roley et al. 2014, Hall et al. 2016) and advances in inverse modeling approaches capable of providing robust estimates of GPP and ER from DO time-series data (Holtgrieve et al. 2010, Grace et al. 2015, Appling et al. 2018*a*). Due to these improvements, there has been an increase in research interest and capabilities focused on stream metabolism (Appling et al. 2018*b*).

Here, we investigated the impacts of urban stream restoration on seasonal and annual nutrient removal within 10 stream reaches in the greater Baltimore, Maryland, USA region. Stream reaches included unrestored urban streams and urban streams restored using a variety of approaches over the past approximately 20 yr. Although specific restoration approaches differed across sites, all restorations included floodplain reconnection and natural channel design, restoring meanders and geomorphic complexity into the study reach. We used 1–2 yr of semicontinuous stream metabolism estimates, coupled with seasonal nutrient releases to quantify nitrate (NO_3^-) spiraling metrics and metabolic activity in restored and unrestored urban stream reaches. We predicted higher metabolic activity (GPP and ER) and more rapid NO_3^- spiraling in restored stream reaches due to improved biotic habitat, and increased water residence times in restored reaches. If stream restoration increases metabolic activity and NO_3^- spiraling, it would suggest that stream restorations could be implemented to reduce NO_3^- export to sensitive downstream water bodies (i.e., the Chesapeake Bay).

METHODS

Study design

We performed this study over a two-year period from April 2015 through April 2017. For the first year of the project, we worked at six total stream reaches: four restored stream reaches and two unrestored stream reaches. In year 2, we continued working at these initial six reaches and added one additional restored reach and three additional unrestored reaches, leading to a total of five restored and five unrestored reaches for the second year of the study. Restored reaches spanned a restoration age gradient, with the oldest restoration being completed in 1999 and the newest restoration being completed in 2014. Most of the reaches are located in small, headwater streams, but we did include two reaches (one restored and one unrestored) with slightly larger watersheds to incorporate stream restorations representing a wider range of stream size (Table 1). Specific restoration practices and objectives varied across reaches, but restorations generally entailed reconnecting the channel with the flood plain, enhancing hydromorphological complexity (i.e., pool–riffle complexes), and reinforcing banks using rip-rap, boulders, and gabion baskets. In addition to restoration status, we categorically grouped reaches by the riparian canopy status (either open or closed canopy), which is an important driver of in-stream nitrogen cycling (Tank et al. 2018). Canopy designation was made qualitatively based on reach surveys during spring and summer periods. Four reaches were selected as paired unrestored-restored reaches (two pairs), with an upstream unrestored study reach and a downstream restored reach. For one of these pairs, the two reaches were immediately adjacent (Scotts Level Branch Tributary), whereas for the other pair, the unrestored reach was ~1,000 m upstream of the restored reach (Scotts Level Branch Mainstem). No major inflows occurred between the two reaches.

Nitrate spiraling quantification

We performed seasonal (spring, summer, autumn) nutrient uptake measurements during baseflow conditions using the short-term plateau nutrient release approach (Tank et al. 2017) to quantify stream nutrient spiraling metrics (Webster and Patten 1979, Newbold et al. 1981, Stream Solute Workshop 1990). If it had rained recently at the site, we waited to perform nutrient releases until nearby (at or downstream of the study reach) USGS streamflow gages on the same stream had returned to baseflow conditions. For most reaches, we performed one year of seasonal releases, but we repeated seasonal releases at one set of paired unrestored-restored stream reaches (Scotts Level Branch Tributary). Upon arriving at each study stream, we delineated an experimental reach (50–125 m), with five sampling locations equally spaced throughout the reach. Prior to setting up the nutrient release, we estimated stream discharge using the mid-section method (Gore and Banning 2017), collected background samples for water chemistry and measured background rhodamine-WT (RWT) concentrations using a Hydrolab MS-5 minisonde (OTT-Hydromet, Loveland, Colorado, USA). For water chemistry samples, we collected 60 mL of site water with a syringe, pushed 20 mL of sample through a 0.7-μm in-line filter (Pall Corporation, Port Washington, New York, USA) into a sterile sample bottle (50-mL

Table 1. Location, restoration age, and watershed information for the 10 study reaches.

Reach	Year of restoration	Latitude	Longitude	Metabolism study period	Spiraling study period	Watershed area (km^2)	Impervious surface cover (%)
Scotts Level Branch Tributary, unrestored (SLBU)	na	39°22′25.84″ N	76°47′41.53″ W	both years	both years	0.7	29.5
Scotts Level Branch Mainstem, unrestored (SLMU)	na	39°22′47.64″ N	76°47′54.24″ W	year 2	year 2	3.0	20.4
Gwynns Falls at Glyndon (GFGL)	na	39°28′18.45″ N	76°49′2.97″ W	year 2	year 2	0.8	19.0
Herring Run (HERR)	na	39°22′24.84″ N	76°35′4.05″ W	both years	year 1	5.9	25.6
Dead Run at Franklintown (DRKR)	na	39°18′40.35″ N	76°43′0.35″ W	year 2	year 2	14.3	39.8
Scotts Level Branch Tributary, restored (SLBR)	2014	39°22′25.58″ N	76°47′39.43″ W	both years	both years	0.8	29.4
Scotts Level Branch Mainstem, restored (SLMR)	2014	39°22′29.57″ N	76°47′35.89″ W	year 2	year 2	3.6	20.6
Minebank Run, older restoration (MBRO)	1999	39°24′7.93″ N	76°34′42.17″ W	both years	year 1	1.0	23.3
Stony Run (STNY)	2005	39°21′20.17″ N	76°37′48.06″ W	both years	year 1	1.8	15.7
Minebank Run, newer restoration (MBRN)	2005	39°24′37.48″ N	76°33′19.84″ W	both years	year 1	5.4	21.3

Note: na, not applicable.

centrifuge tube) as a rinse, shook the rinse, dumped the rinse back into the stream, and then filtered the remaining 40 mL of sample water from the syringe into the sample bottle. We used this process for all water chemistry sampling, using a separate filter, filter housing, and syringe for background samples and for each individual background and plateau sampling station.

After collecting background samples, we made our nutrient release solution by dissolving known masses (or volumes) of NO_3^--N (reactive tracer, as $NaNO_3$), Br^- (conservative tracer, as NaBr), and RWT (added volumetrically using 20% RWT stock). We pre-weighed $NaNO_3$ and NaBr into individual bags based upon mass needed to raise background NO_3^--N and Br^- concentrations by 100 μg/L. Individual bags were portioned out by 5 L/s discharge increments, and we added one bag (each for $NaNO_3$ and NaBr) to the nutrient release solution bucket (20-L bucket, filled with 10 L of site water) for every 5 L/s of discharge, rounding up. For example, if we estimated discharge as 12 L/s, we would add three bags of each pre-weighed salt. We also added the appropriate amount of RWT to raise background RWT concentrations by 20–40 μg/L.

Once the release solution was fully mixed, we added the release solution at a constant, slow rate (target release rate 50 mL/min). We positioned the release solution tubing at the upstream edge of a riffle complex to insure complete mixing by the most upstream sample station. If there were no appropriate riffle complexes, we created one by moving cobbles and boulders to create a mixing zone directly downstream of the tubing. We continuously monitored downstream RWT concentrations using a Hydrolab MS-5 minisonde (OTT HydroMet, Loveland, CO, USA) connecting to a surveyor to provide real-time concentrations. We deemed that plateau had been reached once we observed <5% change in RWT concentrations over a 10-min period. Time to plateau varied widely across reaches, ranging from <0.5 h to >4 h.

Once the stream had reached plateau concentrations, we collected plateau water chemistry samples (triplicate samples from each sampling station) and measured RWT concentrations at the same five sampling locations where we had collected background samples. All samples (plateau and background) were stored on ice until returning to the laboratory, where samples were frozen. Once all plateau samples had been collected, we checked the drip rate of the pump to quantify any fluctuations in pump rate, and then turned off the pump to end the release. After the release was complete, we measured active-channel width at ≥10 transects evenly spaced along the reach and measured ≥5 depths at each of these transects to estimate mean reach width and depth.

All water chemistry samples were returned to the Cary Institute of Ecosystem Studies where they were frozen until analysis. Samples were analyzed for NO_3^--N and Br^- via ion chromatography (Dionex LC20 series ion chromatograph; Thermo Fisher Scientific Inc., Waltham, MA, USA). We calculated NO_3^- uptake length (S_w; m) using an exponential decay model regressing the natural log of the NO_3^- to Br^- ratio against downstream distance from the nutrient release. The inverse of the slope of this model provides an estimate of S_w, which we then converted to uptake velocity (v_f; mm/min) and areal uptake (mg $N \cdot m^{-2} \cdot h^{-1}$) using standard calculations (Stream Solute Workshop 1990, Tank et al. 2017). Any nutrient release that did not yield a significant nutrient uptake result was treated as below our nutrient detection limit. We report the frequency of “below detection” results in the *Results: NO_3^- spiraling* section. For statistical analyses (see *Statistical analyses*), we treat in the Results: NO_3^- spiraling section detection values in a similar manner to chemical analyses: values below detection for nutrient uptake were assigned a value of 0.5× the minimum value from all sites with detectable uptake for v_f and U, and 2× the maximum value of S_w from all sites with detectable uptake. S_w is multiplied because it represents the distance an average molecule travels downstream before being removed, and therefore a longer S_w is interpreted as less nutrient uptake.

Stream metabolism

In addition to quantifying nutrient spiraling seasonally, we deployed dissolved oxygen (DO) and temperature sensors (miniDOT; Precision Measurement Engineering, Vista, California, USA) and light loggers (HOBO pendant logger; ONSET, Cape Cod, Massachusetts, USA) near the downstream end of each reach. We calibrated the light loggers, which quantify light as lux, to estimate PAR by co-locating all loggers with a LI-COR 1300 PAR sensor (LI-COR Biosciences, Lincoln, Nebraska, USA) for one week prior to deployment, and establishing a universal calibration for all sensors (data not shown). Dissolved oxygen, temperature, and light were recorded every 10 min. Sensors were initially deployed at the six reaches where we quantified N uptake in the first year of the project. These sensors remained deployed for the duration of the project, and we deployed an additional set of sensors at the reaches added on during the second year of the project. Two of these reaches (GFGL, DRKR; both unrestored) only have metabolism data available from February 2017 to November 2017 due to technical issues with sensor deployment and data collection. Biofouling of miniDOTs was an issue during early portions of the project, and as a result we have a range of missing data. Once we noticed and diagnosed the biofouling issues, we added anti-fouling copper mesh (supplied by the manufacturer) and began monitoring and cleaning sensors approximately every two weeks throughout the remainder of the project. Furthermore, miniDOTs came out of the water either following a high flow event or during periods of low-water making those data unusable until we performed our regular sensor maintenance and returned the sensors to the channel.

We used the single-station, open-channel O_2 exchange approach (Odum 1956) to estimate stream metabolism.

Our modeling approach followed that of Reisinger et al. (2017), and was based on a modification of the daytime regression approach (Atkinson et al. 2008, Grace et al. 2015). We modeled GPP and ER as

$$[\mathrm{DO}]_{t+1} = [\mathrm{DO}]_t + \mathrm{AI}_t^p - R\left(\theta^{(T_t - T_{\mathrm{mean}})}\right) + K_{\mathrm{O2}} \times \left(1.0241^{(T_t - T_{\mathrm{mean}})}\right) \times \left([\mathrm{DO}]_{\mathrm{sat},t} - [\mathrm{DO}]_{\mathrm{modeled},t}\right) \quad (1)$$

where t is the time step, AI^p is the primary production term (mg $O_2 \cdot L^{-1} \cdot d^{-1}$) where A is a constant, I is surface irradiance, and p is an exponent accounting for photosaturation, R is respiration (mg $O_2 \cdot L^{-1} \cdot d^{-1}$), θ is the temperature dependence of respiration, T_t and T_{mean} are water temperature at time t and average daily water temperature, K_{O2} is the gas exchange coefficient (d^{-1}) and sat and modeled refer to [DO] at saturation and modeled concentrations, respectively. We used the Bayesian single-station estimation (BASE) modeling approach (Grace et al. 2015) modified based on the recommendations of Song et al. (2016) to estimate daily GPP and ER. Although we implemented the BASE model directly in the R statistical software using code available online, the BASE modeling approach is now available through the BASEmetab function in R (R Core Team 2017; code used is *available online*).[6]

Due to high levels of diel temperature fluctuation commonly found in urban streams, we used BASE to simultaneously estimate GPP, ER, K_{O2}, p, and θ. GPP and ER are estimated as volumetric units in base, but in order to compare metabolic rates across reaches, we converted these estimates to areal rates (g $O_2 \cdot m^{-2} \cdot d^{-1}$) by multiplying volumetric rates by mean daily stream depth. For each reach, we established unique depth : discharge relationships using depth measurements made during nutrient releases and nearby USGS discharge gages.

Statistical analyses

Although the initial focus of this study was on the effect of stream restoration on nutrient and energy dynamics of urban streams, preliminary data analysis suggested that variation in stream canopy cover caused by restoration or other riparian vegetation management was a potential driver of nutrient uptake. Therefore, we have included canopy cover in our statistical analyses. Although canopy cover is a continuous variable, it was not an initial focus of our study and we did not quantify canopy cover during the study. Therefore, canopy cover is treated as a binary, categorical variable (open or closed canopy) based on personal observation during the growing season. We tested the effect of stream restoration on NO_3^--N spiraling across all 10 study reaches using ANOVA, with restoration status (restored or not), canopy cover (open or closed), and an interaction term as main effects, season as a block to control for seasonal variation in the nutrient and energy dynamics, and spiraling metrics (S_w, v_f, U) as response metrics. To further isolate the direct effect of stream restoration, while controlling for extrinsic environmental drivers, we performed a separate analysis using only the paired restored-unrestored reaches. For this analysis, restoration status (restored or not) was the main effect and season was included as a block. Both of the restored reaches had open canopies and unrestored reaches had closed canopies, making it impossible to disentangle the direct effect of canopy cover from other restoration effects. However, as opening the canopy was caused by stream restoration activities, if the restoration effect is directly due to an open canopy, it is still ultimately a restoration effect.

Due to the semi-continuous nature of our metabolism data set (missing data due to sensor malfunction or biofouling), we calculated seasonal averages for stream metabolism metrics and N removal estimated via metabolic activity. We averaged all available data from each reach across seasons. We used these seasonal averages as our response metrics and performed the same statistical comparisons (restoration effect, paired reaches, and canopy effect) using GPP, ER, and P:R. Nutrient spiraling metrics were log-transformed prior to running ANOVAs to meet the assumptions of normality and equal variance. We set $\alpha = 0.05$ as our critical value for statistical tests, but we report all P values <0.10.

RESULTS

NO_3^- spiraling

We were able to estimate NO_3^- spiraling for 29 of the 36 NO_3^- releases we performed throughout the study, whereas there was no detectable uptake for seven of the releases (Table 2). Of the releases for which we were able to estimate spiraling metrics, S_w ranged from 10.1 to 697 m, v_f ranged from 0.1 to 66.6 mm/min, and U ranged from 5.0 to 5,077 mg $N \cdot m^{-2} \cdot h^{-1}$. When looking across all study reaches, there was no significant restoration effect on any spiraling metric ($P > 0.05$ for each spiraling metric). Although nonsignificant, there was a potential interaction between restoration status and canopy cover influencing S_w (df = 1,30; $F = 2.998$; $P = 0.093$), with restored sites with open canopies showing a reduction in S_w (increased uptake), and restored sites with closed canopies showing an increase in S_w (reduced uptake; Fig. 1). When focusing solely on paired unrestored-restored reaches, both S_w (df = 1,14; $F = 6.53$; $P = 0.023$) and v_f (df = 1,14; $F = 6.65$; $P = 0.022$) showed higher nutrient spiraling rates (shorter S_w, faster v_f) in restored reaches (Fig. 1). There was also a trend of increased U in restored reaches, although this trend was not significant (df = 1,14;

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[6] https://github.com/dgiling/BASE

TABLE 2. Physicochemical characteristics and nutrient spiraling parameters of study streams.

Season	NO$_3$ (mg/L)	Q (L/s)	Width (m)	Depth (m)	Vel (m/s)	S_w (m)	v_f (mm/min)	U (mg·m^{-2}·h^{-1})
Year 1								
Herring Run, unrestored, open canopy								
Summer 2015	2.22	82.7	1.1	0.17	0.43	609.6	7.14	951.0
Autumn 2015	1.10	11.6	0.8	0.03	0.49	31.1	28.35	1871.3
Spring 2016	1.27	51.0	1.0	0.07	0.73	45.9	66.63	5077.0
Scotts Level Branch Tributary, unrestored, closed canopy								
Summer 2015	0.75	10	2.3	0.19	0.02	319.5	0.84	37.6
Autumn 2015	2.28	1.4	2.4	0.12	0.005	29.4	1.18	161.7
Spring 2016	1.46	1.4	2.2	0.08	0.008	697.0	0.06	5.0
Minebank Run, newer restoration, open canopy								
Summer 2015	1.30	81.4	6.2	0.09	0.15	†	†	†
Autumn 2015	0.86	25.3	6.2	0.09	0.05	190.5	1.29	66.7
Spring 2016	0.76	56.6	6.8	0.11	0.08	236.3	2.10	96.0
Minebank Run, older restoration, closed canopy								
Summer 2015	1.11	22.9	2.6	0.15	0.06	327.0	1.63	108.8
Autumn 2015	0.70	1.26	2.6	0.24	0.002	†	†	†
Spring 2016	1.03	4.15	2.8	0.14	0.01	†	†	†
Stony Run, restored, closed canopy								
Summer 2015	2.18	16.1	2.8	0.09	0.06	283.9	1.22	159.6
Autumn 2015	2.20	4.5	2.5	0.06	0.03	38.8	2.74	361.6
Spring 2016	2.40	8.8	2.7	0.10	0.03	252.0	0.79	113.4
Scotts Level Branch Tributary, restored, open canopy								
Summer 2015	0.75	4.1	2.8	0.08	0.02	83.8	1.03	46.3
Autumn 2015	1.51	1.4	2.7	0.08	0.006	144.9	0.21	19.4
Spring 2016	1.73	1.4	2.8	0.06	0.01	51.7	0.60	62.0
Year 2								
Dead Run at Franklintown, unrestored, closed canopy								
Summer 2016	0.66	33.9	6.1	0.09	0.06	69.1	4.86	192.5
Autumn 2016	3.57	49.0	5.7	0.12	0.07	353.8	1.46	312.3
Spring 2017	3.57	79.0	5.5	0.11	0.13	348.2	2.45	524.4
Gwynns Falls at Glyndon, unrestored, open canopy								
Summer 2016	0.63	0.6	0.8	0.04	0.02	23.7	1.76	66.5
Autumn 2016	2.42	0.5	1.1	0.05	0.01	†	†	†
Spring 2017	7.85	3.9	1.1	0.07	0.05	†	†	†
Scotts Level Branch Tributary, unrestored, closed canopy								
Summer 2016	1.64	7.6	1.9	0.07	0.06	†	†	†
Autumn 2016	12.10	0.5	1.9	0.06	0.004	45.0	0.36	260.2
Spring 2017	13.20	1.1	2.5	0.06	0.007	75.4	0.34	265.5
Scotts Level Branch Mainstem, unrestored, closed canopy								
Summer 2016	0.74	2.2	3.8	0.06	0.01	†	†	†
Autumn 2016	0.53	0.5	3.9	0.05	0.002	17.2	0.45	14.4
Spring 2017	2.06	4.6	4.0	0.04	0.03	35.8	1.90	235.4
Scotts Level Branch Tributary, restored, open canopy								
Summer 2016	1.52	1.8	2.3	0.04	0.02	22.7	2.08	190.1
Autumn 2016	12.02	0.9	2.4	0.06	0.006	12.7	1.71	1235.3
Spring 2017	11.07	1.5	2.0	0.07	0.01	10.1	4.34	2885.3
Scotts Level Branch Mainstem, restored, open canopy								
Summer 2016	0.50	5.8	2.2	0.07	0.04	29.6	5.30	158.9
Autumn 2016	0.53	0.5	1.8	0.07	0.004	36.3	0.45	14.4
Spring 2017	2.56	16.7	2.5	0.09	0.07	22.9	17.35	2664.8

Notes: Physical and chemical conditions and nutrient spiraling parameters during nutrient releases: NO$_3$, ambient NO$_3^-$-N concentrations; Q, measured discharge; width, average reach width; depth, average reach depth; vel., average water velocity in the reach; S_w, nutrient uptake length; v_f, nutrient uptake velocity; U, areal uptake rate.

† Nutrient spiraling releases with no significant uptake are treated as 0.5× the minimum v_f or U value or 2× the maximum S_w value (due to S_w being inversely correlated with nutrient uptake).

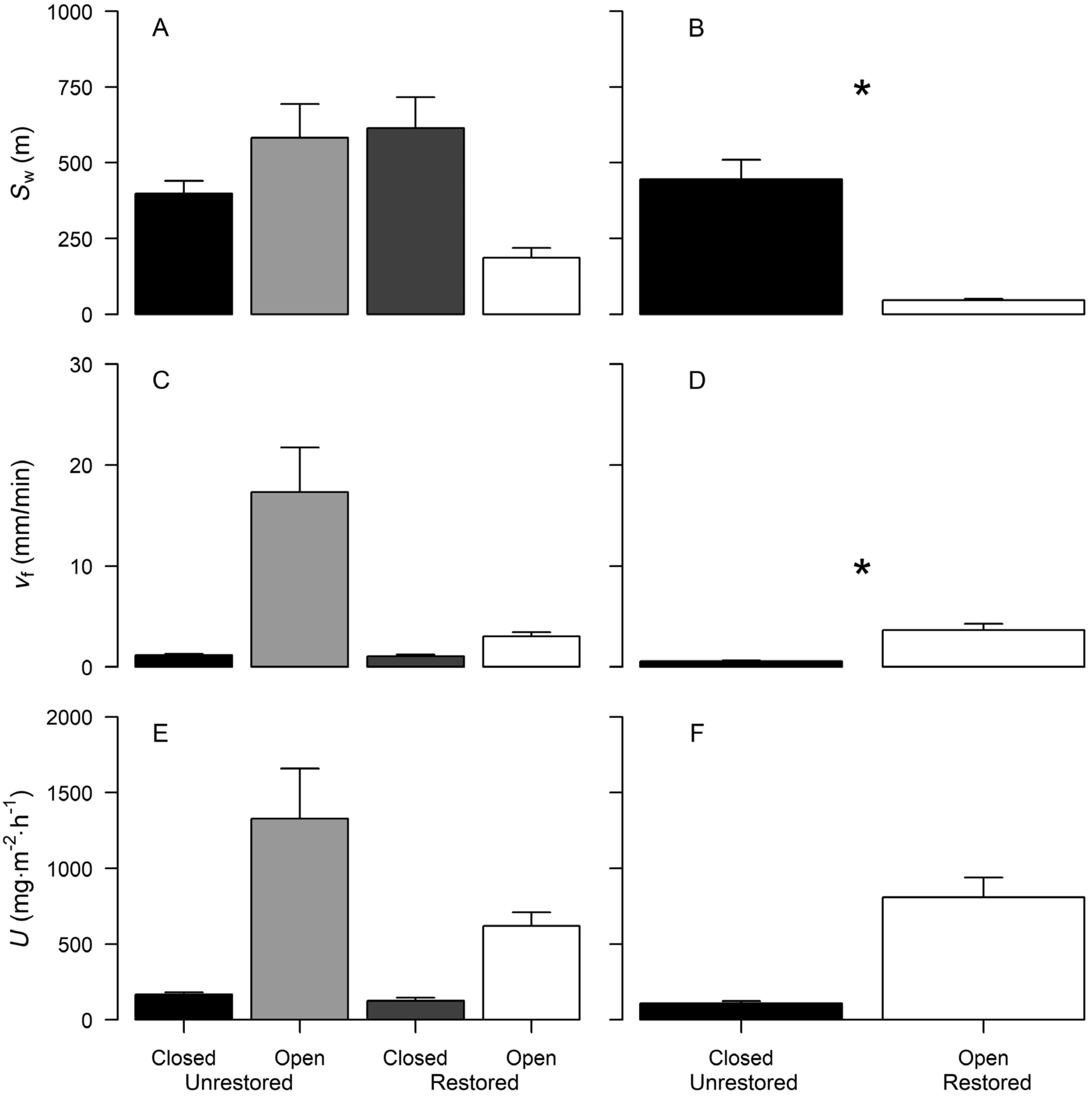

FIG. 1. Nitrate (NO_3^--N) uptake length (S_w; A, B), uptake velocity (v_f; C, D), and areal uptake (U; E, F) averaged across seasons for all reaches (A, C, E) comparing reaches that were closed canopy and unrestored (black), open canopy and unrestored (light gray), closed canopy and restored (dark gray), or open canopy and restored (white) reaches, or comparing paired unrestored and closed canopy (black) and restored and open canopy (white) reaches (B, D, F). Error bars represent one standard error. Asterisks denote significant ($P < 0.05$) effects of restoration or canopy cover, but it is impossible to distinguish the canopy effect from the larger restoration effect for paired reaches as restoration and canopy are perfectly correlated among these four reaches.

$F = 3.61$; $P = 0.078$). There was a non-significant effect of season on S_w in the paired reach analysis (df = 2,14; $F = 3.00$; $P = 0.083$) but no seasonal effects on v_f or U ($P > 0.10$ for each).

Stream metabolism

Stream metabolism equipment was installed for a total of 705 d for the six original stream reaches, which included the final 341 d when the four additional reaches were added to the study. Although equipment was deployed for the entire period of study, various issues (biofouling, sensor failure, low-flow events) produced data gaps throughout our record (Figs. 2 and 3). Across all reaches, we were able to estimate GPP and ER for 61.3% (range 45.5–79.5%) of the total possible days.

Mean GPP was affected by canopy cover (df = 1,33; $F = 5.11$; $P = 0.031$) and there was a non-significant trend of a restoration effect (df = 1,33; $F = 3.28$; $P = 0.079$), but there was no seasonal difference or

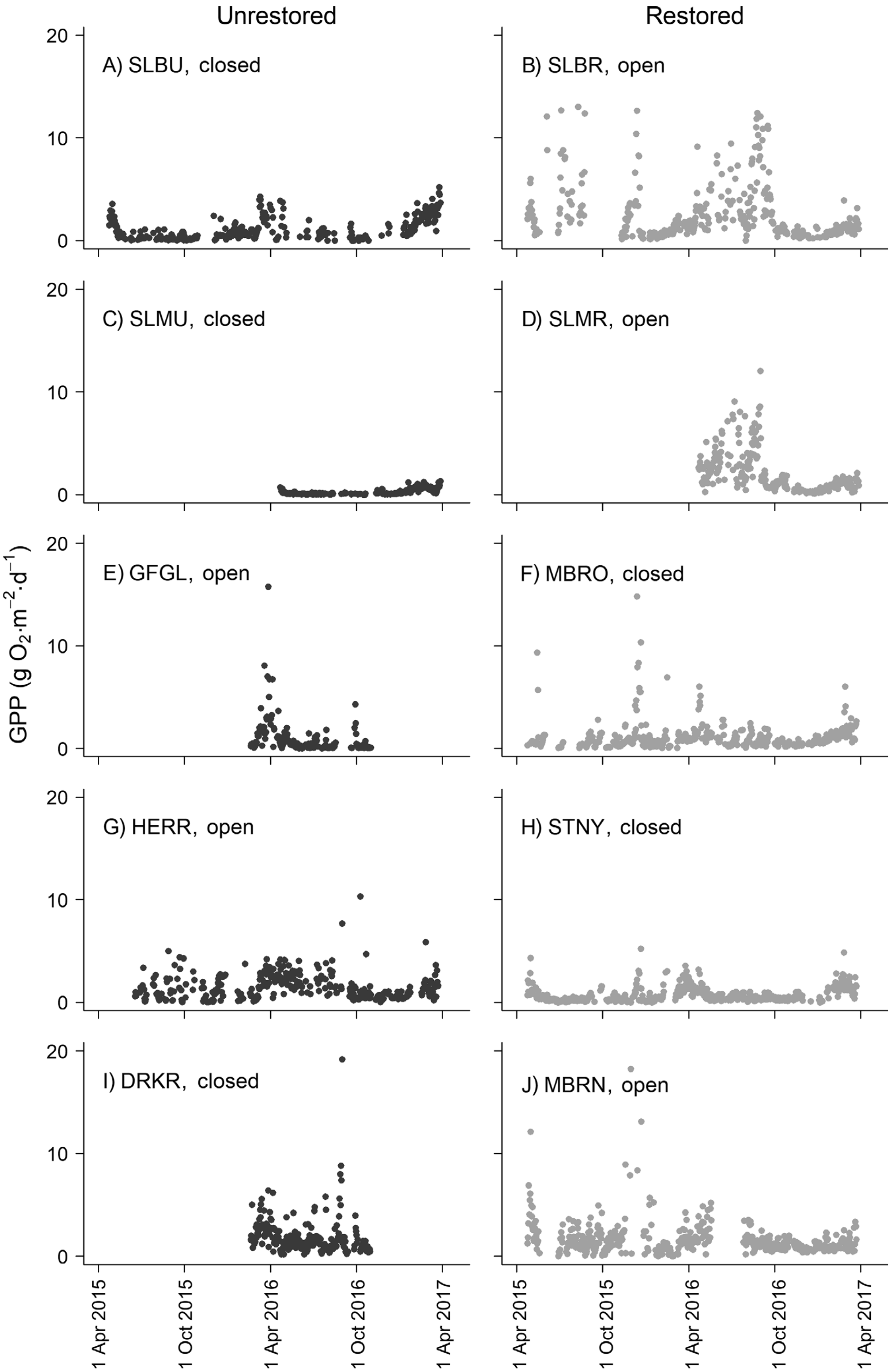

FIG. 2. Gross primary production (GPP) from the unrestored (left panels; dark gray) and restored (right panels; light gray) study reaches included in the two-year study period. Paired (A, B; C, D) unrestored (A, C; dark gray) and restored (B, D; light gray) reaches show the effect (0.010) of restoration or canopy cover (impossible to distinguish among paired reaches; E–J) on GPP. Canopy cover for each reach is denoted along with the reach acronym.

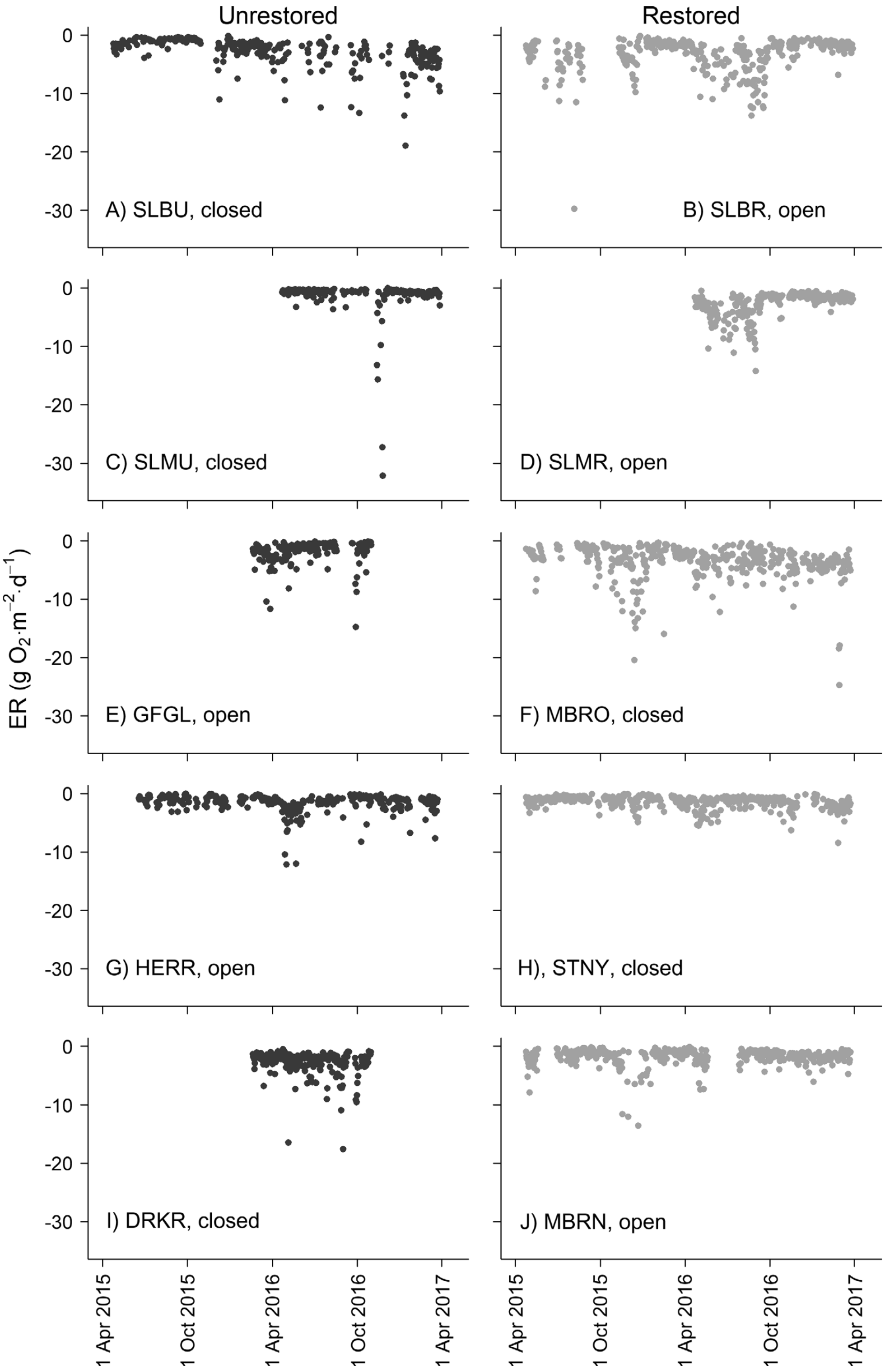

FIG. 3. Ecosystem respiration (ER) from the unrestored (left panels; dark gray) and restored (right panels; light gray) study reaches included in the two-year study period. There was no difference in ER between paired (A, B; C, D) unrestored (A, C; dark gray) and restored (B, D; light gray) reaches, and ER exhibited limited seasonal variability across additional unpaired reaches (E–J) regardless of restoration status. Canopy cover for each reach is denoted along with the reach acronym.

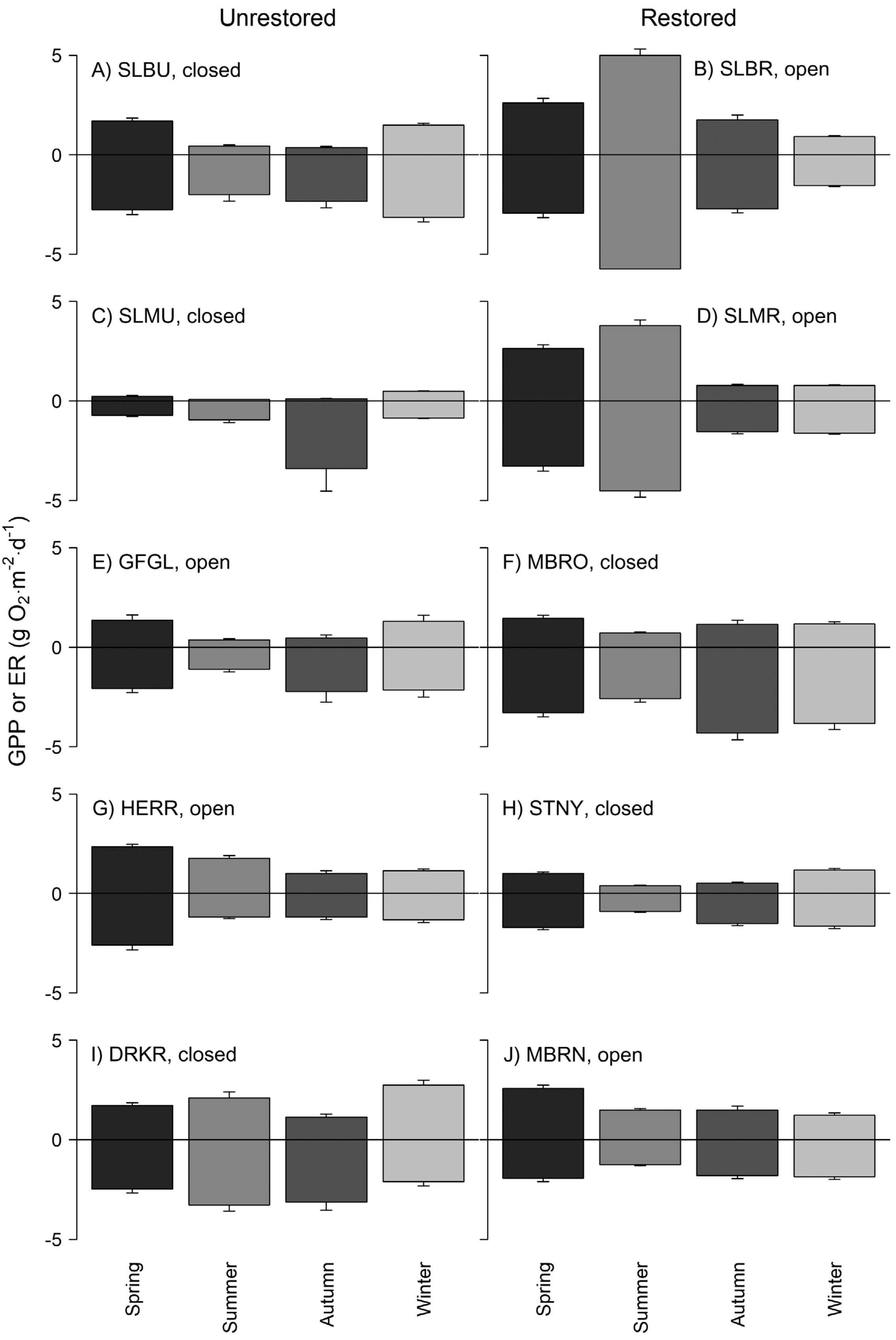
Unrestored
Restored
A) SLBU, closed
B) SLBR, open
C) SLMU, closed
D) SLMR, open
E) GFGL, open
F) MBRO, closed
G) HERR, open
H) STNY, closed
I) DRKR, closed
J) MBRN, open
GPP or ER (g $O_2 \cdot m^{-2} \cdot d^{-1}$)
5
0
-5
Spring
Summer
Autumn
Winter

FIG. 4. Average seasonal gross primary production (GPP; positive bars) and ecosystem respiration (ER; negative bars) across unrestored (left panels) and restored (right panels) reaches in spring (dark fill), summer (light gray), autumn (dark gray), and winter (light fill). Error bars represent one standard error for a given reach–season combination. We note that these error bars use metabolic results for individual modelable days as their level of replication, but this is strictly for visualization purposes. Statistical analyses on metabolism were performed using site as the level of replication. Canopy cover for each reach is denoted along with the reach acronym.

interaction between restoration and canopy cover ($P > 0.10$) (Figs. 2 and 4). In contrast, mean ER was not affected by restoration, canopy cover, their interaction, or season ($P > 0.10$; Figs. 3 and 4). Mean P:R was affected by canopy cover (df = 1,33; $F = 12.27$; $P = 0.001$) but was not affected by restoration, and there was not a significant interaction or season effect ($P > 0.10$; Figs. 4 and 5). Analysis of paired restored-unrestored reaches yielded similar results. Mean GPP of paired reaches was affected by restoration (which includes canopy differences; df = 1,11; $F = 9.64$; $P = 0.010$) but not season ($P > 0.10$), whereas mean ER of paired reaches was not affected by restoration or season ($P > 0.10$). Mean P:R was affected by restoration/canopy cover (df = 1,11; $F = 20.01$; $P < 0.001$) but not season ($P > 0.10$).

When evaluating seasonal P:R patterns (Fig. 5), shifts in seasonality are apparent between open and closed canopy reaches. Closed canopy reaches show strong seasonal patterns, with P:R highest in spring and winter, when leaves are not shading the stream, and lowest in summer and autumn, when leaves prevent light from reaching the stream. This U-shaped pattern on Fig. 5 is evident across the closed-canopy reaches (DRKR, STNY, MBRO, SLBU, SLMU), but less apparent or completely absent in open canopy reaches (GFGL, HERR, MBRN, SLBR, SLMR), which appear to have seasonal P:R patterns driven by temperature, rather than light availability (i.e., highest P:R in spring and summer). Upon initial inspection, it appeared that this seasonality was affected by restoration. However, by further analyzing the data among open and closed canopy streams, the overall pattern appears to be driven by canopy cover rather than restoration (Fig. 5).

DISCUSSION

Urbanization has multiple impacts on stream ecosystems, including altered geomorphology, hydrology, and water chemistry (Walsh et al. 2005), which can lead to economic and environmental impacts due to bank erosion, infrastructure failure, and increased pollutant export. Restorations of urban streams are implemented for a variety of reasons to mitigate these impacts (Bolund and Hunhammer 1999) to protect infrastructure and real estate and address community concerns (Kenney et al. 2012). Although there are often alternative, less expensive options for either retaining nutrients or protecting infrastructure, additional benefits provided by stream restorations (aesthetics, economics, habitat improvement) can overcome the higher costs of restoration. However, the effectiveness of stream restoration for providing these additional benefits, especially nutrient retention, is highly uncertain (Kenney et al. 2012).

In this study, we found that there was no statistical difference in NO_3^- spiraling metrics measured during nutrient release experiments across all sites, but GPP was affected by stream restoration. However, there was a wide range of additional environmental factors that differed across the 10 study reaches. A more focused analysis of the restoration effect based on a pairwise comparison of unrestored and restored reaches within the same stream found that restored reaches had significantly higher NO_3^- uptake (shorter uptake length, faster uptake velocity), although it should be noted that pairwise comparisons were among fairly recent restorations and restored reaches had open canopies whereas unrestored reaches had closed canopies. Therefore, it is not possible to disentangle the effects of restoration and canopy cover with this direct, pairwise comparison. It is likely that this “restoration” effect is driven by opening up of the canopy during the restoration process, mirroring the importance of canopy cover found in a recent review of N spiraling dynamics (Tank et al. 2018). However, even if opening up the canopy was the direct mechanism for increased NO_3^- uptake, the canopy cover was changed due to restoration activity, and therefore restoration was the ultimate driver of this apparent, albeit potentially temporary, effect of reduced canopy cover leading to increased nutrient removal in newly restored urban streams. Furthermore, we found that canopy cover was an important driver of both metabolism dynamics across our 10 study reaches over a two-year monitoring period, which is unsurprising as light availability has been shown to be a significant driver of annual stream metabolism in nearby urban streams, with implications for N concentrations and fluxes (Kaushal et al. 2014, Pennino et al. 2014, Smith and Kaushal 2015).

Urban stream nutrient and energy dynamics

Counter to our expectations, the most biologically active stream in regards to nutrient cycling was Herring Run, a concrete lined stream that is the eighth flashiest (discharge) stream in the United States based on USGS gaging data (E. Doheny, *personal communication*). Although very flashy over the long-term in response to precipitation events, during baseflow conditions the concrete channel provides a stable substrate for algal biofilms to colonize. This stable substrate, coupled with an open canopy and high nutrient concentrations, are the primary mechanisms for exceptionally high rates of NO_3^--N spiraling in Herring Run. A similar result was previously found in Mediterranean streams and diversion canals, with concrete-lined, channelized canals exhibiting higher uptake velocities and areal uptake rates than non-channelized streams (Izagirre et al. 2008).

Although there was no significant effect of stream restorations across our entire set of study reaches, these restorations were highly variable in design, spatial location, and age. This variation may be why previous studies have found no effect of urban stream restoration on metabolic activity (Sudduth et al. 2011), particularly as there can be rapid biogeochemical responses to restoration (Arango et al. 2015), but the restoration effect may decrease over time (McMillan et al. 2014). To control for these variations, we focused specifically on paired unrestored and restored reaches of the same stream to isolate the restoration effect. By conducting pairwise comparisons of restored and unrestored reaches within a stream, we found a significant restoration effect on nutrient uptake velocity, with restored reaches having faster uptake velocities across all seasons. We should note, however, that these paired sets of reaches also represented the newest stream restorations. As these restorations age, the effect on stream metabolism and nutrient uptake may diminish (McMillan et al. 2014). As restorations age and riparian canopies mature, less light reaches the stream, reducing in-stream productivity and altering both energy and nutrient dynamics within these restored stream reaches.

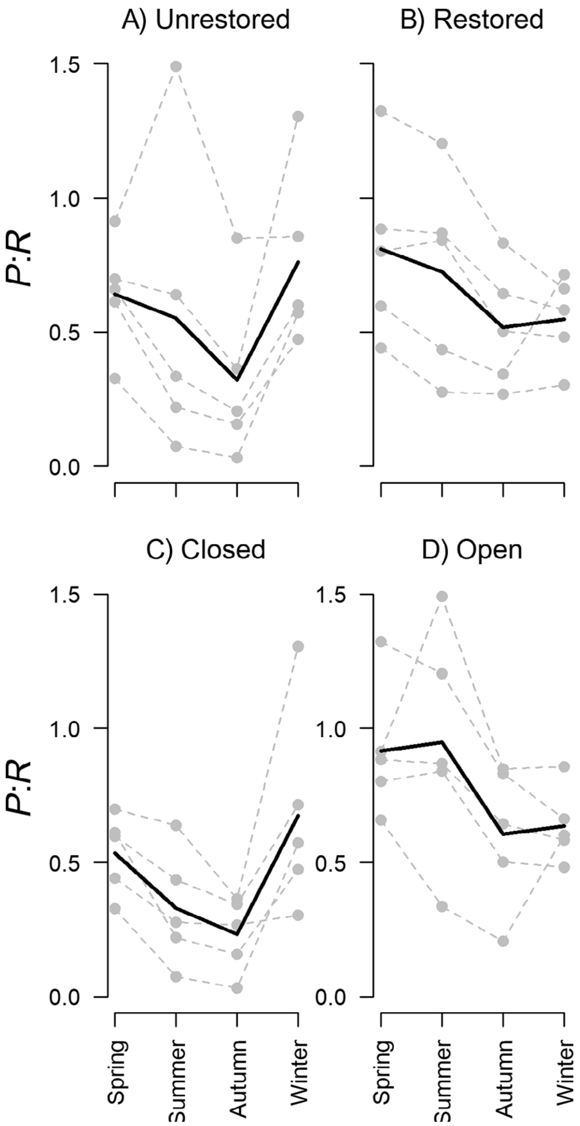

FIG. 5. Seasonal patterns in the productivity to respiration ratio (P:R) differed between (A) unrestored and (B) restored reaches, with unrestored reaches typically showing a U-shaped pattern with spring and winter having higher P:R than summer and autumn, whereas restored reaches typically had highest P:R in spring and summer. These patterns were emphasized when comparing (C) closed and (D) open canopy reaches. Seasonal P:R averages from individual reaches are shown in gray, with the thick solid line representing the average of reaches in each category.

Previous long-term metabolism records have shown that streams are predominantly heterotrophic throughout the year in forested (Roberts et al. 2007), agricultural (Roley et al. 2014), and urban (Duncan et al. 2017) ecosystems. However, urban ecosystems often fluctuate between heterotrophic and autotrophic states, particularly in open canopied urban streams (Alberts et al. 2017). Seasonal patterns in metabolic activity can differ dramatically, driven by changes in light, temperature, and flow regimes (Bernhardt et al. 2018). Finer-scale variation (i.e., days to weeks) in metabolic activity is also evident across a range of streams, but appears to be particularly evident in open-canopied urban streams (this study, Alberts et al. 2017, Reisinger et al. 2017) due to variation in these same factors. Despite large increases in hydrological flashiness associated with the urban stream syndrome (Walsh et al. 2005), urban streams are highly metabolically active, with GPP and ER recovering from flood events in less than 10 days (Reisinger et al. 2017). Because these different environmental factors vary on short timescales, stream metabolic regimes should be viewed at longer (seasonal or annual) scales to compare across sites (Bernhardt et al. 2018).

Seeing the light and the trees in stream restoration projects

Based upon the (semi-)continuous metabolism data, the restoration effect on nutrient and energy dynamics may be driven largely by changes in riparian canopy cover altering the light regime of stream reaches, enhancing autochthonous productivity and N assimilation (Hall and Tank 2003, Tank et al. 2018). This increase in autochthonous production is particularly important for urban streams where natural carbon sources from the watershed may be reduced (Alberts et al. 2017).

Although canopy cover does appear to be driving restoration effects in the current study, the temporal aspect of riparian vegetation must be considered. Early on in the restoration process, light availability drives metabolism and nutrient dynamics (this study, Newcomer Johnson et al. 2014), but as the riparian vegetation community and hydrological connectivity within streams develop, heterotrophic and dissimilatory processes may become more important (Kaushal et al. 2008, Gift et al. 2010, Harrison et al. 2011). There is a need to further address the long-term temporal aspects of stream restoration and cumulative impacts on downstream water quality.

Combining continuous metabolism with seasonal nutrient spiraling measurements revealed that restoration affects the seasonality of metabolism (i.e., altered seasonal P:R patterns), potentially increasing the amount of assimilatory N retention, which can temporarily buffer downstream water bodies from pulses of reactive N. The restoration effect, though, appears contingent upon altered canopy cover, which is likely to be a temporary effect of restoration. As the riparian canopy matures, metabolic regimes appear to return to pre-restoration levels. This is a potentially disturbing result as canopy removal that facilitates nutrient uptake may reduce other ecosystem services (Bolund and Hunhammer 1999), including nutrient processing beyond the stream channel (i.e., riparian or wetland processes; Harrison et al. 2011, Reisinger et al. 2013, McMillan and Noe 2017).

Our results suggest that decision-making about stream restoration, including evaluations of nutrient benefits, clearly need to consider the spatial and temporal dynamics of canopy cover and trade-offs among multiple ecosystem services. Restoration of streams and rivers is a common phenomenon, and restorations are undertaken for different reasons using different strategies (Palmer et al. 2014). Restorations are commonly viewed from a utilitarian perspective, acknowledging the multiple ecosystem services provided by stream restoration beyond the stated objective (Palmer et al. 2014). Our results reiterate that open canopy streams have increased biogeochemical functioning, but to gain a broader understanding of the effectiveness of stream restorations, incorporating these improvements in ecosystem functioning with other restoration objectives (aesthetics, property value improvements, channel stabilization) is needed. Stream restorations perform multiple services for society, and incorporating nitrogen and energy dynamics into the broader picture to gain a utilitarian perspective of stream restoration effectiveness is needed to understand the benefits of these management actions for society.

Acknowledgments

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Data Availability

Data and R scripts used in this publication are available through the Environmental Data Initiative as follows: (1) nitrogen uptake data, https://doi.org/10.6073/pasta/1b3ad75482a940f2ed192fafa0836a05; (2) metabolic activity data, https://doi.org/10.6073/pasta/31c71a726fc07e4ff3dd1163213bf9f2; and (3) raw data input needed for metabolism modeling, https://doi.org/10.6073/pasta/2a230f0c3dba1cf7a3cd1af2c3fb7a3b