Carbon limitation of heterotrophic respiration is linked to dissolved organic matter quality in urban streams

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Abstract

Urban streams are degraded by a suite of factors, including burial beneath urban infrastructure (i.e., roads, parking lots) that eliminates light and reduces direct organic matter inputs to streams, with likely consequences for organic matter metabolism by microbes and carbon limitation in streams. We studied seasonal changes in organic matter metabolism by microbial communities in open and buried reaches of three urban streams in Cincinnati, Ohio. We characterized organic matter quality using fluorescence spectroscopy, extracellular enzyme profiles, and carbon limitation patterns. We hypothesized: 1) that algal production would lead to higher quality dissolved organic matter (DOM) in spring compared to other seasons and in open compared to buried reaches, 2) lower reliance of microbial respiration on recalcitrant carbon sources in spring and in open reaches, and 3) that microbial respiration would be more carbon limited in the autumn and in buried reaches. DOM quality was generally higher in spring than autumn, but the only DOM quality metric that varied by reach was an indicator of recalcitrant humic compounds, which showed more humic DOM in open compared to buried reaches. This likely reflected open reaches as an avenue for direct terrestrial inputs from the riparian zone. Extracellular enzyme assays showed that microbes in buried reaches consistently allocated more effort to degrade recalcitrant carbon sources, consistent with a lack of labile carbon compounds due to limited photosynthesis. Finally, buried and open reaches were both more carbon-limited in autumn when terrestrial leaf inputs dominated compared to the spring when vernal algal blooms were pronounced. Altogether, our data show that stream burial affects the quality of DOM pool with consequences for how microbes use those carbon sources, and that buried and open stream reaches were limited by labile carbon in all seasons. Different carbon quality and use patterns coupled with widespread carbon limitation suggests that these urban streams likely export recalcitrant carbon to downstream water bodies, and that the cycling of nitrogen and/or phosphorus could decrease if heterotrophic metabolism is limited by labile carbon availability.

Introduction

As suburban sprawl converts farmland and forests to urban infrastructure, and as the global trend of urbanization continues (Grimm et al. 2008), the biological function of urban streams and its role in water quality has received increased attention (Kaushal et al. 2015). Relatively small increases in impervious surface cover through urbanization can lead to a “flashy” hydrologic regime that reinforces entrenchment and channel incision in streams that are often already channelized to promote storm water drainage (Dunne and Leopold 1979). These channelized streams are less retentive of particulate organic carbon (Paul and Meyer 2001) which, in combination with greater nutrient loads (Carpenter et al. 1998) and reduced riparian canopies (Griffiths et al. 2013), can alter the contribution of heterotrophic and autotrophic processes to stream metabolism (Kaushal et al. 2014). For example, canopy opening and nutrient enrichment can increase autotrophy (Bernot et al. 2010; Griffiths et al. 2013; Alberts et al. in press), but stream burial can increase the importance of heterotrophy relative to autotrophy (Beaulieu et al. 2014; Pennino et al. 2014). Depending on management, changes in organic matter processing by headwater streams may have an influence on the quantity and quality of organic matter subsidies further downstream along the urban watershed continuum (e.g., Kaushal and Belt 2012, Pennino et al. 2014, Kaushal et al. 2014)

Urban infrastructure expansion frequently results in low order streams being contained in buried pipes (Elmore and Kaushal 2008). Further, open and buried stream reaches often alternate in an urban hydrological network so that stream metabolism can be vastly different in alternating stream reaches. For example, the severe reduction or absence of photosynthetically active radiation (PAR) fundamentally alters a stream ecosystem by eliminating the contribution of primary production to the food web. Although metabolism in buried streams shifts to net heterotrophic conditions, buried streams support a lower overall rate of ecosystem respiration compared to open reaches (Beaulieu et al. 2014; Pennino et al. 2014). Because buried stream reaches are often optimized to convey water quickly and efficiently for drainage purposes, they have increased water velocity which, in conjunction with net reduction in overall biological demand for nutrients (Beaulieu et al. 2014; Pennino et al. 2014), promotes nutrient export to downstream reaches and ecosystems (Beaulieu et al. 2015). Burial also severely affects standing stocks of organic matter in streams, and buried reaches have lower overall coarse and fine benthic organic matter, periphyton, and chlorophyll a standing stocks compared to open reaches (Beaulieu et al. 2014). Organic matter standing stocks in buried reaches also have little seasonality, except for higher coarse benthic organic matter (CBOM) in the fall whereas organic matter standing stock exhibit pronounced seasonal patterns in open streams (Beaulieu et al. 2014). Although the effect of stream burial on particulate organic matter standing stocks has been investigated, how this effect propagates through the microbial community to determine the abundance and quality of dissolved organic matter (DOM) is unknown.

DOM is an important energy source for ecosystem respiration (Meyer and Edwards 1990), and microbial assimilation transfers this energy from dissolved sources to higher trophic levels (Meyer 1994). Streams depend on allochthonous organic carbon inputs from the terrestrial landscape including leaf litter inputs from the riparian zone and DOM exported from soil by groundwater, as well as autochthonous sources from in-stream production of algae and/or macrophytes. These organic matter sources partly determine the quality of the DOM pool available to stream microbial communities. Allochthonous inputs are generally more recalcitrant (i.e., lower quality) than autochthonous sources (McKnight et al. 2001) due to the presence of more structurally complex carbon compounds (e.g., lignin, tannin). In contrast, autochthonous carbon sources have fewer complex structural compounds and relatively more polysaccharides (e.g., cellulose, hemicellulose), so these carbon sources are generally considered more labile (i.e., higher quality). Therefore, the lability of the DOM pool is likely to vary seasonally in conjunction with autumn leaf inputs and vernal algal blooms. Moreover, urban infrastructure likely also affects the DOM pool composition with open reaches having more labile DOM than buried reaches due to greater light availability and associated higher levels of primary production (Kaushal et al. 2014), and greater hyporheic exchange and higher respiration rates in open reaches (Beaulieu et al. 2014) could influence microbial use and transformation of DOM. These seasonal and reach-scale differences in organic matter dynamics in urban streams are likely to influence the quality of the organic carbon pool and associated rates of microbial carbon processing.

We used a nutrient diffusing substratum (NDS) approach coupled with extracellular enzyme activity (EEA) assays and characterized DOM via fluorescence to understand how organic carbon demand varies seasonally in buried and open stream reaches of urban streams. EEA assays characterize how microbes allocate energy to acquire different compounds (e.g., labile or recalcitrant carbon, nitrogen, etc.). These assays quantify the microbial demand for and environmental availability of substrates (Sinsabaugh and Follstad Shah 2012), and they have been used to infer microbial organic nutrient limitation patterns in soils and sediments (e.g, Sinsabaugh et al. 2009) and within river networks (Hill et al. 2012). DOM fluorescence properties can characterize various fractions of DOM as more or less labile, and allochthonous or autochthonous. This technique has been used to investigate seasonal (Catalan et al. 2013) and landscape (Williams et al. 2016) differences in the composition of the DOM pool in surface waters. We defined three hypotheses based on anticipated seasonal and spatial patterns of organic matter and light availability in urban streams containing open and buried reaches. We hypothesized that spring would have higher quality DOM than other seasons due to higher algal production prior to leaf-out of riparian trees, warming stream temperatures, higher sun angle, and longer day length, and that open reaches would have higher quality DOM than buried reaches due to more algal production in open versus buried reaches. Consequently, we hypothesized that microbes in spring would produce lower extracellular enzyme indicators associated with recalcitrant carbon acquisition, and that microbes in open reaches would exhibit less effort to acquire recalcitrant carbon compared to those in buried reaches. Finally, we hypothesized that microbial respiration would be more carbon-limited in autumn due to the pulse of low quality terrestrial organic matter from the riparian zone, and that buried reaches would be more carbon-limited than open reaches due to lower primary production and lower inputs and less retention of allochthonous carbon inputs. Regardless of reach or season, we predicted that microbial respiration would respond more strongly to higher quality carbon amendments in the NDS compared to lower quality carbon amendments.

Methods

Study Sites and Experimental Design

We studied three urban streams in or near Cincinnati, Ohio (USA) consisting of paired buried and open study reaches separated by a 30-100 m buffer reach. Two buried reaches flowed through corrugated pipe and one through a concrete tunnel, and buried stream widths ranged from 0.5-4.5 m. Open reaches were generally incised with restricted riparian zones, contained mobile sandy sediments, and ranged in width from 2.1-3.9 m. In two of the three streams, the buried reach was upstream of the open reach. A more detailed site description can be found in Beaulieu et al. (2014).

We collected water samples to characterize dissolved organic matter quality in summer and autumn 2011 and in spring 2012. Concurrently, we collected biofilms from unglazed clay tiles that were deployed in the streams for a minimum of 6 weeks for extracellular enzyme activity analysis. Microbial carbon limitation patterns were measured using nutrient diffusing substrata. This design allowed us to compare how carbon quality, microbial enzyme activity, and the biofilm response to added carbon varied in space (buried versus open stream reaches) and time (summer, autumn, and spring). We also collected a suite of other environmental data including water chemistry, hydrologic parameters, organic matter standing stocks, and whole stream metabolism and nitrate (NO3-) uptake to understand how those factors related to the microbial response to variations in DOM quality. Nitrate uptake and hydrologic parameters (i.e., transient storage) were measured using whole-stream 15N-NO3- and bromide (Br-) releases. Methods that describe the processing of isotope samples, calculating NO3- uptake rate, and modeling transient storage parameters and one- and two-station whole-stream metabolism are beyond the scope of this paper, but are detailed in Beaulieu et al. (2014).

DOM Characterization

Dissolved organic matter quality was characterized using fluorescence excitation-emission matrices (EEMs; Coble et al. 1990, Coble 1996, Cory et al. 2010) measured on a Fluoromax-4 spectrofluorometer (Horiba Instruments, Kyoto, Japan). This technique quantifies humic-like, fulvic-like, and protein-like fractions within the bulk DOM pool, which in turn are generally related to the lability or recalcitrance of DOM pool. EEMs were measured using excitation wavelength at 10 nm intervals between 240-450 nm at and emission wavelengths at 2 nm intervals from 290-600 nm. Three-dimensional EEMs were then instrument corrected, blank subtracted, and normalized by the water Raman signal (Cory et al. 2010) using Matlab software, but we did not measure absorbance for each sample, so we could not perform the standard inner-filter correction on the EEMs. Therefore these results will be most useful for relative differences across sites and time rather than for comparison to literature values.

The EEMs were used to calculate several DOM quality indices, including the humification index (HIX; Zsolnay et al. 1999; Huguet et al. 2009), the biological freshness index (BIX; Huguet et al. 2009), the fluorescence index (FI; McKnight et al. 2001), and the protein-to-humic ratio (P/H; Coble 1996; Stolpe et al. 2010). HIX characterizes the humic or autochthonous fractions of DOM (Zsolnay et al. 1999; Ohno 2002), and it is calculated as the ratio of integrated fluorescence emission intensity between 300-345 nm and between 435-480 nm at 254 nm excitation. Higher HIX values indicate DOM with humic character whereas lower values indicate either less humic or more autochthonous DOM. BIX was calculated from the ratio of emission at 380 and 430 nm at excitation of 310 nm (Huguet et al. 2009). BIX values <0.7 are associated with allochthonous DOM, values 0.8-1.0 are associated with autochthonous DOM, and values >1.0 are associated with aquatic bacterial sources; higher values indicate greater lability than lower values. FI is calculated from the ratio of the fluorescence intensity at 450 nm and 400 nm at excitation of 370 nm. FI values of about 1.9 indicate fulvic acids from microbes and values of about 1.4 indicate terrestrial-origin fulvic acids. Finally, P/H was calculated from the EEMs whereby excitation at 275 nm and emission at 340 nm is associated with protein-like organic matter and excitation at 350 and emission at 480 is associated with humic-like organic matter (Coble 1996; Stolpe et al. 2010).

Extracellular enzyme activities (EEA)

Biofilm collected from tiles deployed in the buried and open reaches was analyzed for extracellular enzyme activities (EEA). Microbial assemblages produce extracellular enzymes to degrade organic matter and to acquire nutrients from their environment, and the activity of those enzymes serves as an index of environmental resource availability (Sinsabaugh and Foreman 2001). Acquisition of labile carbon compounds was measured as -D-glucosidase activity, and acquisition of recalcitrant carbon compounds was measured as polyphenol oxidase (POX) and peroxidase activity. An alternate metric of recalcitrant carbon acquisition was measured as the activity of L-3,4-dihydroxyphenylalanine (DOPA) + H2O2 as a substrate, and this metric correlates with lignin degradation. The ratio of recalcitrant carbon acquisition to total carbon acquisition (as -D-glucosidase + polyphenol oxidase) characterizes the overall quality of the DOM pool (equivalent to lignocellulose index or LCI) whereby values greater than 0.5 indicate greater effort to acquire recalcitrant carbon and values less than 0.5 indicate greater effort to acquire labile carbon (Sinsabaugh and Follstad Shah 2011). We also used CQI, the ratio of labile to recalcitrant carbon acquisition enzymes whereby larger values indicate greater effort to acquire labile carbon, as alternate metric of overall carbon quality. Nitrogen acquisition was measured as the activity of -N-acetylglucosaminidase (NACE: EC 3.2.1.50).

All EEA assays used microplate protocols (Sinsabaugh et al. 1997; Sinsabaugh and Foreman 2011) modified by Hill et al. (2010). Microplate arrays were run with quadruplicate assays for each tested enzyme and reference standard, which were prepared in sterile deionized water. Fluorescence quenching, or the decrease of emissions caused by interaction between target enzyme substrates and non-reactant chemicals, was measured by comparing fluorescence of standard solutions mixed with sample to that of standard solution mixed with buffer. We measured fluorescence (Model FLX800T, BioTek Instruments, Winooski, VT, USA) at excitation wavelength of 350 nm and emission wavelength of 450 nm.

Nutrient diffusing substrata (NDS)

NDS arrays were deployed throughout the open reaches, and where light was extinguished at the upstream and downstream ends of the buried reaches. We initially predicted increased carbon limitation at the downstream end of each buried reach due to microbial processing of DOM through the buried reach, but we found no difference in carbon-limitation or EEM metrics between the upstream and downstream ends of the buried reaches. Therefore upstream and downstream NDS arrays were both considered “buried” in the statistical analysis. Each NDS array had one of four 0.5 M carbon amendments (glucose, arabinose, cellobiose, or a no-carbon control (n=8 each)) to represent increasing recalcitrance. The NDS were supplemented with 0.5 M N as NH4Cl and 0.5 M P as KH2PO4 to alleviate any potential nutrient limitation that could confound interpretation of the heterotrophic response to added carbon, and we used porous glass disks rather than cellulose sponges to eliminate the heterotrophic response to the sponge as a particulate carbon source. NDS arrays were installed within open-ended PVC for shade to reduce the potential for autotrophic biofilms to colonize the glass disks. NDS arrays were collected after a two week deployment and shipped overnight on ice for laboratory analysis within 24 h.

Laboratory analysis for biofilm respiration consisted of submerging the NDS disks in site water, incubating the disks in the dark for 3.5 h, and recording net oxygen change from the start to the end of the incubation. The glass disks were saved for calculation of biomass after weighing oven-dried (60 °C) samples before and after combustion in a muffle furnace (500 °C). The respiration response was scaled by disk area (g O2 cm-2 h-1) and by biomass (mg O2 gAFDM-1 h-1), and in order to compare the respiration response among streams and seasons, we calculated the nutrient response ratio (NRR) as respiration response for an individual NDS cup divided by the mean control response for that particular deployment.

Water chemistry and hydrologic parameters

We collected filtered (0.45 m) water samples in the field and stored them on ice for transport to the laboratory where they were acidified or frozen depending on the analyte. We used standard colorimetric methods to measure nitrate + nitrite (hereafter, NO3-), dissolved reactive phosphorus (DRP), ammonium (NH4+), and bromide (Br-) on a flow injection analyzer (Lachat Instruments, Loveland, CO USA). Dissolved organic carbon (DOC) concentration was measured using high-temperature Pt-catalyzed combustion and NDIR detection (Shimadzu TOC-VCPH, Columbia, MD, USA).

The breakthrough curve of Br- released in conjunction with the 15N-NO3- release was used in OTIS-P (Runkel, 1998), a one-dimensional advection, dispersion and transient storage model, to estimate solute hyporheic exchange parameters such as the cross-sectional area of the transient storage zone (As), the storage zone exchange coefficient (),the storage zone residence time (Tsto), and the storage exchange flux (qs) fraction of the median travel time due to transient storage, F200med (Runkel, 2002). These methods are described in detail in Beaulieu et al. (2014).

Organic matter standing stocks

We collected 10-20 samples of organic matter from different habitat units in a stratified-random sampling design. Samples for coarse (>1 mm), fine (<1 mm), and attached (i.e., periphyton) organic matter were collected from 0.052 m2 isolated by an open-ended plastic cylinder placed no more than 5 cm into the sediment. Coarse benthic organic matter (CBOM) was removed by hand, and the sediments were agitated before taking a fine benthic organic matter (FBOM) subsample. We collected periphyton by scraping a known area (0.006-0.04 m2) of a rock with a wire brush. We calculated sample dry mass and ash-free dry mass of samples by weighing oven-dried (60 °C) samples before and after combustion in a muffle furnace (500 °C). We used a subsample of periphyton to measure chlorophyll a using the trichromatic method (APHA 2005) following hot ethanol extraction (Sartory and Grobbelaar 1984).

We deployed unglazed clay tiles for six weeks at all sites to provide a standardized surface for algae and bacteria to colonize. Biofilm on tiles was removed with a toothbrush and razor blade, rinsed into a bottle with site water, and stored on ice until analysis. Subsets were analyzed for algal abundance using a Palmer-Maloney counting cell (Charles et al. 2002), total bacterial counts using qPCR, and extracellular enzyme activity assays. Detailed methods for these analyses are described in Beaulieu et al. (2014).

Statistical Analysis

We used multivariate generalized least squares linear models (GLS) with alternate variance structures and model optimization (Zuur 2009) to test how DOM quality (HIX, BIX, FI, P/H) differed among seasons (summer, autumn, spring) and between reaches (buried, open). We also used GLS to test for differences in extracellular enzyme activity (POX, DOPA-H2O2, LCI, NACE) and carbon limitation patterns among seasons and between reaches. We examined the relationship between CQI and LCI using Spearman’s rank correlation. We used linear modeling to test relationships between carbon limitation patterns and water chemistry, hydrologic parameters, organic matter standing stocks, and whole stream metabolism and NO3- uptake. We used permutational multivariate analysis of variance using distance matrices (adonis in the vegan package for R; Oksanen et al. 2016) to detect a relationship between the aggregated response of microbial respiration to glucose, arabinose, and cellobiose NDS additions and CBOM and FBOM standing stocks. Other parameters were not significantly related to the NDS data. All statistical analyses were done using R (R Core Team 2016)

Results

Patterns in DOM Variability

DOM quality differed among seasons (summer, autumn, spring) and between reaches (buried, open). HIX, the humification index, differed by season (GLS, p=0.0005), with autumn having higher HIX than spring or summer, which were not different from each other. HIX also differed by reach (GLS, p=0.021) with open reaches having higher HIX than buried reaches when compared across all seasons (Figure 1). Because we did not perform the standard inner-filter corrections on these samples, these values show relative differences between reaches and among seasons.

BIX, the biological freshness index, and FI, the fluorescence index, varied by season (GLS, p<<0.0001) but did not differ between buried and open reaches (Figure 2A and 2B, respectively). Although BIX and FI did not differ between spring and summer, both indices had significantly lower values in autumn compared to spring and summer. The BIX and FI values we measured indicate low autochthonous content and terrestrially-derived fulvic acids respectively in all seasons and reaches.

P/H, the protein to humic ratio was generally <1 indicating high humics. This ratio varied by season (GLS, p<<0.001), with spring and summer having a higher ratio (more protein) compared to fall (Figure 3), and also by reach (GLS, p<<0.0002) with open reaches having lower ratio (more humic-like) than buried reaches when all seasons were combined.

Patterns in extracellular enzyme activity

Although differences in DOPAH2 activity among seasons were not detected, DOPAH2 activity was higher in biofilm from buried reaches than in biofilm from open reaches (GLS, p=0.024) when we expressed DOPAH2 per unit dry mass (Figure 4a) or per unit carbon (data not shown). Polyphenol oxidase (POX) extracellular enzyme activity within biofilm was higher in buried reaches compared to open reaches (GLS, p=0.0043) (Figure 4b).

We found no evidence of spatio-temporal differences in extracellular enzyme activity (EEA) associated with labile carbon use. However, biofilm LCI values from buried reaches reflected higher use of recalcitrant carbon than open reaches (GLS, p=0.014), and summer biofilm had greater use of recalcitrant carbon than autumn biofilm (GLS, p=0.027). There were no differences between spring and autumn (Figure 5). The LCI was also correlated to the CQI (rs=-0.98, p<<0.0001, data not shown).

Because carbon uptake and use is often linked to the acquisition of N from the environment, we also analyzed differences in N uptake as activity of -N-acetylglucosaminidase. We measured highest values in the autumn, intermediate values in summer, and lowest values in spring with all seasons significantly different from each other (GLS, p<<0.0001) (Figure 6), but there were no differences between open and buried reaches.

Carbon limitation

We deployed NDS amended with different carbon sources (glucose, arabinose, cellobiose, and a no-carbon control) to see if patterns in carbon limitation differed between buried and open stream reaches or among seasons. The NDS we deployed during summer were washed away by stormflows. Therefore, we focus our analysis on autumn and spring to contrast the carbon limitation response to a time when leaf inputs dominate compared to when vernal algae blooms dominate.

Respiration rates on NDS disks were not different among carbon source treatments when the data were scaled by biomass (mg O2 gAFDM-1 h-1). However, when the respiration response was scaled by disk area (g O2 m-2 h-1), all NDS carbon amendments were significantly different than the control in all streams, seasons, and reaches (GLS, p<<0.001). Respiration response were not detectably different among the three carbon amendments during any deployment (GLS, p>0.05). Generally, fall had higher NRR (ratio of the treatment response to the control) compared to spring in both reaches (LME, p<<0.0001; Figure 7). We found a significant interaction (GLS, p=0.0009) between season (autumn versus spring) and reach (buried versus daylight) whereby the respiration response to added carbon was stronger for open compared to buried reaches in autumn, but it was stronger for buried compared to open reaches in spring (Figure 7). Further, the difference between the seasonal responses was less pronounced in buried reaches than in open reaches. Overall, these results indicate carbon limitation in all streams and season.

No relationships between NRR and water chemistry, hydrology, or ecosystem-scale functional attributes were detected. Although EEA and DOM quality metrics often differed between seasons and reaches, there was no direct linear relationship between NRR and those metrics. Although most standing stock metrics were also unrelated to NRR, there were weak positive relationships between reach-scale standing stocks of CBOM (adonis, p=0.036) and FBOM (adonis, p=0.053) with glucose, arabinose, and cellobiose NRR.

Discussion

Seasonal patterns of DOM characteristics

These urban streams had higher CBOM biomass in autumn compared to other seasons and higher chlorophyll a biomass in spring than in other seasons (Figure 8). Because terrestrial carbon sources are typically of lower quality than aquatic autochthonous DOM sources (McKnight et al. 2001), these changes in CBOM and chlorophyll biomass should result in lower quality DOM dominating in autumn and higher quality DOM dominating in the spring. As we hypothesized, BIX and FI, metrics of labile DOM, show a clear pattern of less labile carbon during autumn and more labile carbon during spring, likely due to riparian leaf fall producing a large influx of recalcitrant terrestrial DOM in the fall and vernal algal blooms producing a large influx of labile autochthonous DOM in the spring. HIX, which measures the recalcitrant humic fraction of DOM, is similar to BIX and FI with autumn having higher humic character than spring or summer. This pattern is also seen in the P/H (protein/humic) ratio, which shows more humic-like components in the autumn compared to the spring whereas summer was not distinctly different. Collectively, these patterns reflect the reach-scale standing stock data collected during this study. This seasonal pattern is seen in temperate forested mountain streams (Villanueva et al. 2016), ephemeral Mediterranean streams that flow during the autumn-spring wet season (Catalan et al. 2013), and in other urbanized streams (Hosen et al. 2014). Therefore, temperate zone seasonality of autumn riparian leaf inputs and spring algal blooms imparts the dominant seasonal signature to the DOM pool of these temperate urban streams even though they have limited riparian zones due to channelization.

Despite the strong and consistent seasonal differences measured across multiple DOM optical properties and higher primary production and algal standing stocks in the spring, the low absolute values of BIX and FI show that the DOM pool in the seasons we studied has a weak autochthonous component and a strong signature of terrestrially-derived fulvic acids. The dominance of terrestrial or humic derived carbon in the DOM pool may be a general pattern in streams draining urbanized basins. For example, microbial humic-like DOM compounds were associated with higher population density and greater proportion of developed land across nearly 200 catchments in southeast Canada (Williams et al. 2016). In contrast to the higher microbial humic signature in Williams et al. (2016), we found more terrestrial humic-like DOM compounds instead, suggesting a terrestrial DOM source. Terrestrial DOM sources include upwelling ground water, leaking stormwater infrastructure (Kaushal and Belt 2012), and runoff from impervious surfaces (Hope et al. 2014). DOM derived from these sources may overwhelm any autochthonous signature in streams. Alternatively, the year-round stronger, more recalcitrant terrestrial characteristics could indicate that heterotrophic biofilms, which are typical in urban streams (Johnson et al. 2009), rapidly remove high quality DOM from the water column. For example, Franke et al. (2013) found that labile autochthonous carbon stimulated water column carbon use for energy metabolism and/or assimilation. Furthermore, the presence of algal biofilms enhanced the EEA of heterotrophic biofilms, suggesting the rapid use of labile DOM in the presence of autochthonous production (Rier et al. 2014). Rapid use of high quality DOM would be consistent with systemic carbon limitation, which we found in all reaches and seasons.

Spatial patterns of DOM characteristics

Our hypothesis that open reaches would have more labile carbon than buried reaches was not supported by the optical properties of the DOM pool. Although there were lower overall chlorophyll and CBOM biomass in buried compared to open reaches, reach type was not a significant predictor of BIX or FI, metrics that indicate labile DOM. One explanation for the lack of a burial effect on BIX and FI is that these optical properties of the DOM pool are determined by processes at the larger stream segment or catchment scale, rather than the reach scale. For example, in a previous study at the stream network scale and across a range of discharges in urbanized catchments, BIX never had a strongly autochthonous character despite many instances of net ecosystem productivity in the spring across 30 months of continuous sampling (Smith and Kaushal 2015). Further, a cross-system study found that catchment scale land use was a good predictor of DOM composition (Williams et al. 2016), which implies that catchment urbanization could have overwhelmed reach-scale differences in organic matter dynamics in our highly urbanized streams (16-34% impervious surface cover; Beaulieu et al. 2014).

In contrast to BIX and FI, which were not affected by burial, HIX (an indicator of humic DOM) was higher in open reaches compared to buried reaches, which was contrary to our hypothesis that buried reaches would have lower quality DOM. This pattern was also reflected in the P/H (protein-to-humic) ratio, which was likely driven by the relative abundance of humic-like compounds (denominator of the ratio) rather than patterns in aquatic production that affected low molecular weight fractions of the DOM pool (numerator of the ratio), consistent with the year-round humic nature of DOM in these urban streams. The pattern of higher HIX in open reaches was largely driven by high HIX in autumn (Figure 1) when open reaches received and retained more leaf litter that could leach recalcitrant terrestrial DOM (Figure 8). In contrast, buried reaches neither received direct inputs of riparian leaf litter nor retained litter exported from upstream due to higher velocities and fewer retention structures (Beaulieu et al. 2014). Alternatively, dilution of the DOM pool by lower HIX sewage sources that leak into the buried reaches (Smith and Kaushal 2015) could reduce HIX in the buried reaches. Sorption of humic compounds has been observed in other studies (Ohno 2002; Zsolnay et al. 1999), but we collected water for EEM metrics from the top and bottom of the buried reaches, and there was no significant difference in HIX collected at either end of a buried reach (data not shown). Although the EEA data indicated greater use of recalcitrant carbon in buried reaches compared to open reaches (see below), the lack of change in HIX as water flows through the buried reaches implies that microbial processing of humic compounds was not enough to reduce the HIX of the DOM pool. Higher HIX in the open reach in spring is counter-intuitive given the presence of large algal standing stocks and high GPP, which would be expected to produce labile DOM. It is possible that the high HIX values resulted from DOM leached from greenfall inputs during leaf out and/or flower or seed production (Lewis and Likens 2007), and although we did not observe a CBOM peak in spring, greenfall leaches at higher rates than senescent leaves (Fonte and Schowalter 2004). However, in the overall context of this study, the median spring HIX values are lower than autumn in open and buried reaches, which is still consistent with an overarching seasonal affect driven by terrestrial sources in the autumn and aquatic sources in the spring.

Despite spatio-temporal differences in the DOM composition driven by seasonal differences in CBOM and algae, the DOM data show that these urban streams are dominated by terrestrial humic sources, likely from constant seepage of DOM from soils to streams throughout the watershed. These data also show secondary control over DOM quality due to spatial differences in organic matter inputs that alter the characteristics of the DOM pool in buried versus open reaches. Therefore urban infrastructure can influence the characteristics of the DOM pool in the urban stream network.

Patterns in Carbon Use – Extracellular Enzyme Activities (EEA)

Extracellular enzyme activity (EEA) reflects the composition of the DOM pool, as perceived by the microbial community. Although P/H and HIX indicated more humic and recalcitrant DOM in open reaches, buried reaches had higher DOPAH2 and POX activity (indicators of recalcitrant carbon) than open reaches. This supports our hypothesis that the microbial community in buried reaches would allocate more energy toward acquiring recalcitrant carbon sources than in the open reach, regardless of season. This pattern is consistent with experiments showing greater POX activity in low light conditions (Wagner et al. 2015). Lower values of DOPAH2 and POX in the open reach indicate less effort to acquire recalcitrant carbon, likely because DOM leached from primary producers supplies an alternative, high quality carbon source. Conversely, the greater effort to acquire recalcitrant carbon in buried reaches is consistent with low chlorophyll, limited periphyton cover, and an extremely low reach-scale GPP (reported in Beaulieu et al. 2014). This implies rapid use of high quality carbon produced in the open reaches and little export to downstream buried reaches, and is consistent with generally greater EEA in the presence of algal biofilms (Rier et al. 2014).

The LCI, which aggregates several EEA measures into a composite index of carbon use, also shows greater use of recalcitrant carbon in buried reaches. However, LCI shows a seasonal effect whereby summer has greater use of recalcitrant carbon than autumn, but that autumn and spring were not different. This pattern may be driven by low CBOM, low chlorophyll a, and high FBOM in open reaches during the summer, and it suggests that the autumn pulse of terrestrial CBOM leaches a labile fraction of DOM that microbes can use despite being dominated by low BIX and FI compounds. Furthermore, the lack of difference in enzyme activity between spring and autumn despite the major differences in CBOM and chlorophyll may reflect the overall terrestrial signature of the DOM pool, which is dominated by terrestrial sources even in the spring.

While the spatial patterns in EEA are consistent with our hypotheses, EEA patterns do not match patterns in the optical properties of the DOM pool. This discrepancy may be due to differences in the composition of the DOM pool in the water column, where the EEM samples were collected, and at the sediment-water boundary layer where the microbial community was sampled for EEA. For example, labile carbon produced in the benthos of the open reaches may be rapidly and selectively processed by microbes with little being transported to the water column. Alternatively, the discrepancy between the optical and microbial indicators may be due to a temporal mismatch. The microbial EEA indicators likely reflect the integrated response of the microbial community to a DOM pool that varies on a diurnal basis with primary production, whereas the optical indices reflect the composition of the DOM pool at the moment the grab sample was collected. The mismatch between EEA and EEMs might also be related to specific substrates used in the EEA assays not corresponding to the compounds that determine the optical properties of DOM.

Although some EEA metrics did not conform to the DOM characteristics, others did. In our streams, N-acquiring enzymes had the lowest abundance in the spring, coincident with higher quality algal DOM, and highest values in summer and autumn, when overall chlorophyll is low and the system is dominated by lower quality FBOM and CBOM standing stocks respectively. Greater N-acquiring activity is associated with increasing C recalcitrance (Sinsabaugh and Follstad Shah 2012), so this finding is consistent with a more labile pool of carbon in spring and a more recalcitrant pool in other seasons. For example, organic matter C:N ratio was lower during spring in forested Mediterranean streams (Villanueva et al. 2016), and higher quality spring DOM in temperate rainforest streams was likely used as a source of labile C and N (Fellman et al. 2009). We found no significant seasonal differences between NO3- or NH4+ concentrations (reported in Beaulieu et al. 2014), suggesting that higher quality spring DOM acted as a nitrogen source as well as a carbon source. Previous work has found seasonal changes in microbial demand for organic N in response to changes in C:N ratio and composition of organic matter (Kaushal and Lewis 2005), and more work needs to be done to understand the role of organic matter as an energy source vs. a nitrogen source in some urban streams. The combined approach of using EEA and EEMs provides complementary information about the characteristics of, and microbial use of, the DOM pool, and the combine approach confirms that spatio-temporal differences in the DOM pool, driven in part by urban infrastructure, translate to spatial differences in how microbes use carbon sources in the urban stream network.

Patterns in Carbon Use – NDS

Biofilms in autumn were always more carbon-limited than in spring, which supported our hypothesis that terrestrial leaf fall would depress DOM quality in autumn. However, the pattern of carbon limitation by reach (i.e., buried or open) varied among seasons. Open reaches were more strongly carbon-limited than buried reaches in autumn, but were less carbon-limited than buried reaches in the spring. Stronger carbon limitation in open reaches during autumn may be a result of the pulse of recalcitrant DOM from terrestrial leaves that entered the open reaches during leaf-fall whereas lower carbon limitation in open reaches during spring may be a result of the pulse of labile DOM derived from algal sources. These explanations are corroborated by the DOM optical properties. Total DOC concentration did not vary between seasons (data not shown), suggesting that the pulse of autumn leaves and spring algae blooms changed DOM composition rather than quantity.

Alternatively, differences in C limitation between reaches might be related to secondary reach-scale factors. For example, EEA assays confirm that biofilms in buried reaches always invested more effort to acquire recalcitrant carbon, so they might have been better able to utilize the autumn pulse of terrestrial DOM compared to the open reaches. In contrast, biofilms in open reaches always invested less in recalcitrant carbon acquisition which, when compounded by the fact that the pulse of autumn leaves was delivered directly to the open reaches, could have led to more intensified carbon limitation. Similarly, in the spring, open reaches responded less to the simple carbon sources in the NDS because the system had higher levels of high quality algal DOM, but P/H ratio shows that buried reaches appear to receive less of this higher quality DOM, so they responded more strongly to the NDS. Less high quality DOM exported to buried reaches is consistent with the potential for rapid use of algal DOM *in situ* by heterotrophic biofilms (Franke et al. 2013; Rier et al. 2014) and is reflected in the carbon acquisition effort devoted to recalcitrant carbon sources.

We found different results when we expressed carbon limitation by area or biomass (i.e., gAFDM-1). When expressed by area, the temporal and spatial patterns were highly significant, but no patterns were evident when expressed by biomass. Therefore, the biofilm response to added carbon is not to increase the per cell carbon use rate, but simply to accumulate greater biomass. Given the fact that we relieved N and P limitation to focus on the carbon amendment response, these results might be most applicable to agricultural and urban streams which tend to have chronically high background nutrient concentrations (Carpenter et al. 1998). The rapid processing of added carbon could also be a function of generally high inorganic nutrient concentrations in these urban streams in combination with the nutrients added to the NDS (Rosemond et al. 2015).

Although we hypothesized that responses to the different carbon types in the NDS arrays would vary, biofilms responded similarly to all carbon sources (glucose, arabinose, cellobiose). Although arabinose has been used as a less labile form of carbon in some studies (e.g., Newbold et al. 2006), our results show that it is just as bioavailable as glucose in this study system. Similarly, we used cellobiose as a breakdown product of cellulose that we predicted would be less bioavailable than glucose or arabinose, yet it was equally bioavailable to those more simple carbon sources. It is unclear if arabinose and cellobiose bioavailability is equally high as glucose in most streams or if it was high in these urban streams because of the systemic dominance of recalcitrant carbon and/or the presence of N and P in the NDS agars.

Overall, these results indicate spatio-temporal variation in biofilm carbon use patterns related primarily to seasonal changes in the DOM pool and secondarily to reach scale patterns, such as stream burial. We found that the pulse of labile autochthonous carbon in the spring might have acted as a nutrient source as well as an energy source, but more work is needed to resolve this conclusively. Additionally, we documented widespread carbon limitation in these urban streams which could have been induced by the dominance of recalcitrant OM sources from the watershed, limited *in situ* production of labile DOM due to stream burial, high background nutrient concentration leading to rapid CBOM consumption (e.g., Rosemond et al. 2015), or some combination of those factors. Together, differences in carbon use patterns among buried and open reaches likely has implications at the river network scale, particularly in drainages dominated by urban infrastructure that alternate between buried and open stream reaches. Because the limited quantity of labile carbon is more likely to be used *in situ*, urban systems with buried reaches may export a higher proportion of recalcitrant carbon than unburied streams, possibly increasing C flux from streams to receiving water bodies, but reducing labile carbon subsidies. Further, when DOM sources are dominated by recalcitrant carbon, uptake and use of nitrogen and phosphorus could decrease, further loading downstream ecosystems with nutrients. Therefore, differential carbon use along the urban stream continuum is likely to have consequences for biogeochemical cycling of other nutrients and for downstream export of DOM, nutrients, and inorganic carbon.

Our work also suggests important considerations for management and restoration of urban streams. Stream daylighting is an engineering approach to urban stream restoration whereby buried streams are redesigned to be open to light (Pinkham 2000). Daylighting is increasingly seen as an effective management approach to improve stream water quality and ecosystem function with respect to nutrient cycling in urban ecosystems (Beaulieu et al. 2015; Pennino et al. 2015). Our results show that an additional mechanism of improvement may be to increase high quality autochthonous labile organic carbon availability to microbes that can support nitrogen removal processes such as denitrification (Newcomer et al. 2012). Few studies have examined the biogeochemical impacts of daylighting streams (Newcomer Johnson et al. 2016) and how the ecosystem changes over time after daylighting. Future research on carbon limitation in buried streams should elucidate how daylighting affects stream ecosystem function and provisioning of ecosystem services like nutrient reduction, especially in the broader context of watershed carbon sources and floodplain connectivity.

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Citations

Alberts, J.M, Beaulieu, J.J., Buffam I. In Press. Watershed land-use and seasonal variation constrain the influence of riparian canopy cover on stream ecosystem metabolism. Ecosystems DOI: 10.1007/s10021-016-0040-9.

APHA (2005) Standard methods for the examination of water and wastewater. American Public Health Association, Washington DC

Beaulieu, J. J., P. M. Mayer, S. S. Kaushal, M. J. Pennino, C. P. Arango, D. A. Balz, T. J. Canfield, C. M. Elonen, K. M. Fritz, B. H. Hill, H. Ryu, and J. W. Santo Domingo. 2014. Effects of urban stream burial on organic matter dynamics and reach scale nitrate retention. Biogeochemistry 121:107-126.

Beaulieu JJ, HE Golden, CD Knightes, PM Mayer, SS Kaushal, MJ Pennino, CP Arango, DA Balz, CM Elonen, KM Fritz, and BH Hill. 2015. Urban stream burial increases watershed-scale nitrate export. PLOS One DOI:10.1371/journal.pone.0132256

Booth DB, Jackson CR. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. J. Am.Water Resour. Assoc. 33:1077–90

Catalan N, Obrador B, Alomar C, and Pretus JL. 2013. Seasonal and landscape factors drive dissolved organic matter properties in Mediterranean ephemeral washes. Biogeochemistry 112:261-274.

Charles DF, Knowles C, Davis RC (2002) Protocols for the analysis of algal samples collected as part of the U.S. Geological Survey National Water-Quality Assessment Program. In: Report No. 02-06. Patrick Center for Environmental Research, The Academy of Natural Sciences, Philadelphia, p 124

Coble. P. G. 1996. Characterization of marine and terrestrial DOM in seawater using excitation emission matrix spectroscopy. Marine Chemistry 51(4):325–346

Coble P. G., S. A. Green, N.V. Blough, and R. B. Gagosian. 1990. Characterization of dissolved organic-matter in the black-sea by fluorescence spectroscopy. Nature 348(6300):432–435

Cory RM, Miller MP, McKnight DM, Guerard JJ, Miller PL. 2010. Effect of instrument-specific response on the analysis of fulvic acid fluorescence spectra. Limnology and Oceanography Methods 8:67–78

Dunne T, Leopold LB. 1978.Water in Environmental Planning. New York: Freeman. 818 pp.

Elmore AJ and SS Kaushal. 2008. Disappearing headwaters: patterns of stream burial due to urbanization. Frontiers in Ecology and Environment 2008 6:308-312

Fellman JB, Hood E, D’Amore DV, Edwards RT, White D. 2009. Seasonal changes in the chemical quality and biodegradability of dissolved organic matter exported from soils to streams in coastal temperate rainforest watersheds. Biogeochemistry 95:277-293

Fonte SJ and Schowalter TD. 2004. Decomposition of greenfall vs. senescent foliage in a tropical forest ecosystem in Puerto Rico. Biotropica 36:474-482.

Franke D, Bonnell EJ, and Ziegler SE. 2013. Mineralisation of dissolved organic matter by heterotrophic stream biofilm communities in a large boreal catchment. Freshwater Biology 58:2007-2026.

Griffiths NA, JL Tank, TV Royer, SS Roley, EJ Rosi-Marshall, MR Whiles, JJ Beaulieu, and LT Johnson. 2013. Agricultural land use alters the seasonality and magnitude of stream metabolism. Limnology and Oceanography 58:1513-1529.

Grimm NB, Faeth SH, Golubiewksi NE, Redman CL, Wu J, Bai X, and Briggs JM. 2008. Global change and the ecology of cities. Science 319:756-760

Hansen AM, Kraus TEC, Pellerin BA, Fleck JA, Downing BD, Bergamaschi BA. 2016. Optical properties of dissolved organic matter (DOM): effects of biological and phytolytic degradation. Limnology and Oceanography 61:1015-1032.

Hill BH, McCormick FH, Harvey BC, Johnson SL, Warren ML, Elonen CM (2010) Microbial enzyme activity, nutrient uptake and nutrient limitation in forested streams. Freshw Biol 55(5):1005–1019

Hill BH, CM Elonen, LR Seifert, AA May, and Ellen Tarquinio. 2012. Microbial enzyme stoichiometry and nutrient limitation in US streams and rivers. Ecological Indicators 18:540-551

Hope, D, Naegeli MW, Chan AH, Grimm NB. (2004). Nutrients on asphalt parking surfaces in an urban environment. Water, Air, and Soil Pollution 4:371-390.

Hosen JD, McDonough OT, Febria CM, Palmer MA. 2014. Dissolved organic matter quality and bioavailability changes across an urbanization gradient in headwater streams. Environmental Science and Technology 48:7817-7824.

Huguet A, Vacher L, Relexans S, Saubusse S, Froidefond JM, Parlanti E (2009) Properties of fluorescent dissolved organic matter in the Gironde Estuary. Organic Geochemistry 40(6):706–719

Johnson LT, JL Tank, and WK Dodds. 2009. The influence of land use on stream biofilm nutrient limitation across eight North American ecoregions. Canadian Journal of Fisheries and Aquatic Sciences 66:1081-1094

Kaushal SS and Lewis WM. 2005. Fate and transport of organic nitrogen in minimally disturbed montane streams of Colorado, USA. Biogeochemistry 74:303-321.

Kaushal, SS and KT Belt. 2012. The urban watershed continuum: evolving spatial and temporal dimensions. Urban Ecosystems 15:409-435.

Kaushal, S.S.; Delaney-Newcomb, K.; Findlay, S.E.G.; Newcomer, T.A.; Duan, S.W.; Pennino, M.J.; Sivirichi, G.M.; Sides-Raley, A.M.; Walbridge, M.R.; Belt, K.T. Longitudinal patterns in carbon and nitrogen fluxes and stream metabolism along an urban watershed continuum. Biogeochemistry 2014, 121, 23–44.

Kaushal SS, McDowell WH, Wollheim WM, Newcomer Johnson TA, Mayer PM, Belt KT, and Pennino MJ. 2015. Urban evolution: the role of water. Water 7:4063-4087.

Klein RD. 1979. Urbanization and stream quality impairment. Water Resour. Bull. 15:948–63

Lewis GP and Likens GE. 2007. Changes in stream chemistry associated with insect defoliation in a Pennsylvania hemlock-hardwoods forest. Forest Ecology and Mangement 238:199-211

McKnight DM, Boyer EW, Westerhoff PK, Doran PT, Kulbe T, Andersen DT (2001) Spectrofluorometric characterization of dissolved organic matter for indication of precursor organic material and aromaticity. Limnology and Oceanography 46(1):38–48

Meyer, JL. 1994. The microbial loop in flowing waters. Microbial Ecology 28:195-199.

Meyer, JL and RT Edwards. 1990. Ecosystem metabolism and turnover of organic carbon along a blackwater river continuum. Ecology 71:668-677

Newcomer, Tamara A., Sujay S. Kaushal, Paul M. Mayer, Amy R. Shields, Elizabeth A. Canuel, Peter M. Groffman, and Arthur J. Gold. 2012. Influence of natural & novel organic carbon sources on denitrification in forested, degraded-urban, & restored streams. Ecological Monographs 82:449–466

Newcomer Johnson TA, SS Kaushal, PM Mayer, RM Smith, and GM Sivirichi. 2015. Nutrient Retention in Restored Streams and Rivers: A Global Review and Synthesis. Water 8:166, doi:10.3390/w8040116.

Ohno T (2002) Fluorescence inner-filtering correction for determining the humification index of dissolved organic matter. Environmental Science and Technology 36(4):742–746

Jari Oksanen, F. Guillaume Blanchet, Roeland Kindt, Pierre Legendre, Peter R. Minchin, R. B. O'Hara, Gavin L. Simpson, Peter Solymos, M. Henry H. Stevens and Helene Wagner (2016). vegan: Community Ecology Package. R package version 2.3-5. https://CRAN.R-project.org/package=vegan

Paul MJ and JL Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333-365

Pennino, M.J.; Kaushal, S.S.; Beaulieu, J.J.; Mayer, P.M.; Arango, C.P. Effects of urban stream burial on nitrogen uptake and ecosystem metabolism: Implications for watershed nitrogen and carbon fluxes. Biogeochemistry 2014, 121, 247–269.

R Core Team (2016). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Regnier P, Friedlingstein P, Ciais P et al. 2013. Anthropogenic perturbation of the carbon fluxes from land to ocean. Nature Geoscience 6:597-607.

Rier ST, Shirvinski JM, and Kinek KC. 2014. In situ light and phosphorus manipulations reveal potential role of biofilm algae in enhancing enzyme-mediated decomposition of organic matter in streams. Freshwater Biology 59:1039-1051.

Rosemond AD, JP Benstead, PM Bumpers, V Gulis, JS Kominoski, DWP Manning, K Suberkropp, and JB Wallace. 2015. Experimental nutrient additions accelerate terrestrial carbon loss from stream ecosystems. Science 347:1142-1145

Runkel R.L. (1998) One-Dimensional Transport with Inflow and Storage (OTIS): A Solute Transport Model for Streams and Rivers. U.S. Geological Society, Water Resources Investigations Report 98-4018.

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

Sartory DP, Grobbelaar JU (1984) Extraction of chlorophyll a from freshwater phytoplankton for spectrophotometric analysis. Hydrobiologia 114(3):177–187

Sinsabaugh RL, Findlay S, Franchini P, Fisher D (1997) Enzymatic analysis of riverine bacterioplankton production. Limnol Oceanogr 42(1):29–38

Sinsabaugh RL, Foreman CM (2001) Activity profiles of bacterioplankton in a eutrophic river. Freshwater Biol 46(9): 1239–1249

Sinsabaugh RL, BH Hill, JJ Follstad Shah (2009) Ecoenzymatic stoichiometry of microbial organic nutrient acquisition in soil and sediment. Nature 462:795-798.

Sinsabaugh RL, Follstad Shah JJ (2011) Ecoenzymatic stoichiometry of recalcitrant organic matter decomposition: the growth rate hypothesis in reverse. Biogeochemistry 102(1–3):31–43

Sinsabaugh RL, Follstad Shah JJ (2012) Ecoenzymatic stoichiometry and ecological theory. Annual Review of Ecology, Evolution, and Systematics 43:313-343.

Smith RM and Kaushal SS. 2015. Carbon cycle of an urban watershed: exports, sources, and metabolism. Biogeochemistry 126:173-195.

Stolpe B, Guo L, Shiller AM, Hassellov M (2010) Size and composition of colloidal organic matter and trace elements in the Mississippi River, Pearl River and the northern Gulf of Mexico, as characterized by flow field-flow fractionation. Marine Chemistry 118(3–4):119–128

Villanueva VD, Navarro MB, Albarino R. 2016. Seasonal patterns of organic matter stoichiometry along a mountain catchment. Hydrobiologia 227-238.

Wagner K, Besemer K, Burns NR, Battin TJ, Bengtsson MM. 2015. Light availability affects stream biofilm bacterial community composition and function, but not diversity. Environmental Microbiology 17:5036-5047.

Williams CJ, Frost PC, Morales-Williams AM, Larson JH, Richardson WB, Chiandet AS, and Xenopoulos MA. 2016. Human activities cause distinct dissolved organic matter composition across freshwater ecosystems. Global Change Biology 22:613-626

Zuur AF, Ieno EN, Walker NJ, Saveliev AA, and Smith GM. 2009. Mixed effects models and extension in ecology with R. Springer, New York, NY.

Zsolnay A, Baigar E, Jimenez M, Steinweg B, Saccomandi F (1999) Differentiating with fluorescence spectroscopy the sources of dissolved organic matter in soils subjected to drying. Chemosphere 38(1):45–50

Tables

Table 1. Coefficients from adonis (Oksanen et al. 2016), a permutational multivariate analysis of variance using distance matrices, show weak relationships between nutrient response and particulate carbon standing stocks

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | Glucose NRR | Arabinose NRR | Cellobiose NRR | *P*-value |
| CBOM | 0.072 | 0.060 | 0.064 | 0.036 |
| FBOM | 0.014 | 0.011 | 0.01 | 0.053 |

Figure Captions

Figure 1. Spatio-temporal variation in the humification index (HIX) values derived from excitation-emission matrices. Lines within boxes are medians, box ends are 1st and 3rd quartiles, whiskers are 1.5 times the interquartile range.

Figure 2. Seasonal variation in the (A) biological freshness index (BIX) and (B) fluorescence index (FI) values derived from excitation-emission matrices.

Figure 3. Spatio-temporal variation in the protein-to-humic ratio (P/H) values derived from excitation-emission matrices.

Figure 4. Reach-scale variation in (A) L-3,4-dihydroxyphenylalanine (DOPA) + H2O2 (DOPAH2) and (B) polyphenol oxidase (POX) activities.

Figure 5. Spatio-temporal variation in the lignocellulose index (LCI) values, and index of carbon quality where larger values indicate more recalcitrant carbon in the dissolved organic matter pool.

Figure 6. Seasonal variation in -N-acetylglucosaminidase (NACE) activity of stream biofilms.

Figure 7. Spatio-temporal variation in the nutrient response ratio (NRR: respiration/mean control) to added carbon when measured on an areal basis (g O2 cm-2 h-1).

Figure 8. (A) Coarse benthic organic matter (CBOM) standing stocks, (B) fine benthic organic matter (FBOM) standing stocks, (C) benthic chlorophyll *a*, and (D) periphyton standing stocks in the buried and open reaches during each sample season. All error bars are standard errors of the mean. Figure originally in Beaulieu et al. 2014.

Figures

Figure 1.

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Figure 2.

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Figure 5.

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Figure 6.



Figure 7.

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Figure 8.

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