

Riparian subsidies and hierarchical effects of ecosystem structure on leaf breakdown in Appalachian coalfield constructed streams



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ABSTRACT

Stream construction is a compensatory mitigation strategy that is commonly used in the Appalachian coalfield, eastern USA, to offset surface-mining impacts to headwater streams. Mitigation assessments currently rely on structural metrics, and there is a paucity of research concerning ecosystem functions in streams constructed for mitigation purposes on mined areas. For eight such constructed streams and four reference streams in the coalfields of southwestern Virginia, we evaluated catchment- and riparian-level land cover, measures of stream ecosystem structure, and riparian litterfall and in-stream leaf breakdown functions across two years. Mean litterfall to reference streams was approximately four times greater than constructed-stream levels. Leaf breakdown rates in coarse-mesh bags were typically faster in reference streams, with reference means exceeding constructed-stream means by 40–50%. Depressed constructed-stream rates resulted from diminished microbial + physicochemical processing of leaves and reduced macroinvertebrate-mediated breakdown. However, breakdown rates in some constructed streams were similar to reference-stream means. Among constructed streams, leaf breakdown rates varied negatively with detention-pond coverage and stream temperature and varied positively with riparian-forest cover. Our results suggest hierarchical relationships among these factors and that rapid establishment of woody riparian canopy could accelerate restoration of organic-matter functions in constructed headwater streams.

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1. Introduction

Surface mining in the Appalachian coalfields of the eastern USA is a major driver of the regional economy and a primary agent of land-use change (Townsend et al., 2009). In the central Appalachians, where the primary geomorphic activity is high-intensity surface mining, estimated rates of geomorphic change caused by human activity exceed rates at which earth is moved by rivers or anthropogenic agents in other U.S. regions (Hooke, 1999). Landscape alteration can affect lotic ecosystems (Allan, 2004), and central Appalachian surface mining is impacting aquatic resources (USEPA, 2005, 2011). Because of the region's highly dissected land-

scape and relatively high drainage densities (Leopold et al., 1964), headwater streams are particularly prone to mining impacts.

Coal surface-mining disturbs extensive areas through soil and rock excavation, valley fill (VF) construction, and other activities. From 1992 through 2002, the U.S. Environmental Protection Agency (USEPA) documented that VFs and associated mining activities caused a direct “loss” of an estimated 1900 km of headwater streams, representing approximately 4% of 1st- and 2nd-order streams in the central Appalachian coalfields (USEPA, 2005, 2011). Headwater streams account for more than 70% of total stream length in the continental U.S. (Leopold et al., 1964) and are unique ecosystems that enhance biodiversity of riverine systems (Meyer et al., 2007), link soil, surface, and groundwater biogeochemical processes (McClain et al., 2003; Vidon et al., 2010), provide landscape hydrologic connectivity (Freeman et al., 2007), and transport matter and energy from riparian zones down river networks (Vannote et al., 1980; Hill and Webster 1983; Wallace et al., 1995a).

Headwater streams disturbed by central Appalachian mining typically originate in forested landscapes. The energetic resource

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base in small forested streams is largely allochthonous organic matter (OM) derived from riparian vegetation (e.g., Fisher and Likens, 1973; Cummins, 1974; Vannote et al., 1980; Iversen et al., 1982; Hill and Webster 1983; Wallace et al., 1995b; Hall et al., 2000). Coarse particulate OM is converted into fine particles via coupled biotic and abiotic processes, and some of this OM is assimilated as biomass by primary consumers contributing to secondary production. Additionally, riparian subsidies as woody debris can increase retention of particulate OM (Webster et al., 1990, 1994) and fine sediments (Sutherland et al., 2002; Valett et al., 2002), and concomitantly, restrict nutrient export (Webster et al., 2000; Valett et al., 2002) and influence community structure (Wallace et al., 1995a; Palmer et al., 1996). Considering the extent of mining impacts to headwater streams (Palmer et al., 2010) and the importance of headwaters to stream networks, need for effective mitigation is evident.

The compensatory mitigation rule, established under section 404(b)(1) of the Clean Water Act (CWA), mandates mitigation of stream losses (33 CFR § 332/40 CFR § 230). Construction of new stream channels on mined areas is a common means of mitigation in central Appalachian coalfields. Post-construction monitoring and assessment is often required as a condition of CWA section 404 permits. Although the CWA requires that mitigation projects replace lost stream-ecosystem structure and function, assessment protocols have routinely relied on visual habitat evaluations (Palmer and Hondula, 2014) or have incorporated other structural measures such as water chemistry, channel stability, and specific biotic assemblage metrics (e.g., Sparks et al., 2003). However, regulators are placing increased emphasis on integrating functional measures into mitigation assessment protocols (Harman et al., 2012).

We quantified rates of riparian litterfall and leaf breakdown for perennial streams constructed on southwestern Virginia mine sites as mitigation for mining disturbances (constructed streams) and in natural perennial headwater streams (reference streams) during two years. Study objectives were to: (1) compare structural attributes and OM functions (riparian inputs and leaf breakdown) for constructed- and reference streams, (2) determine if individual constructed streams functioned similarly to reference streams, and (3) investigate structural attributes in constructed streams that were associated with OM functions to identify factors that may foster functional restoration in constructed streams.

2. Methods

2.1. Site selection and description

We studied eight low-order streams constructed on mine sites and four forested reference streams in southwestern Virginia on similar geology (Fig. 1). These streams are located within the central Appalachians ecoregion (Level III, ecoregion 69; Omernik, 1987). Sites were selected such that basic stream and catchment geomorphic attributes were similar between the two site types (Table 1). For each study stream, we delineated 100-m reaches with consistent channel morphology and riparian structure. During July 2010, we performed (1) physical habitat evaluations according to USEPA Rapid Bioassessment Protocols (RBP; Barbour et al., 1999), (2) field surveys following the methods of Fritz et al. (2006), and (3) examination of stream and catchment characteristics using aerial photographs to aid in site selection.

Constructed streams were located on mining-influenced lands; all were established as distinct channels with mine spoil serving as bank or bed substrates. Construction techniques ranged from rudimentary channel excavation to Natural Channel Design (NCD; USDA-NRCS, 2007) coupled with riparian plantings. Seven constructed streams were intended as compensatory mitigation and

the eighth (CRI) was established as a purposeful effort to reproduce appearance and stability of a natural stream channel (Table 1).

Reference streams represent relatively undisturbed headwater conditions, had physical habitat scores ≥ 175 (Barbour et al., 1999), and mature riparian forests on each bank ≥ 50 m wide. Point-source discharges, stream crossings, residences, and commercial development were not present in any reference catchment upstream of the study-reach.

2.2. Stream, riparian, and catchment geospatial characterization

We quantified geomorphic and land-cover characteristics of study streams (ArcGIS 10.1; Esri, Inc., Redlands, CA). Digital elevation models (DEMs) created from Virginia Base Mapping Program's digital terrain data (VBMP, for 2011, <http://www.vita.virginia.gov/isp/default.aspx?id=12118>; Futrell and Sforza, 2004) were used to delineate catchments. We used a combination of VBMP orthoimagery (2011), features from the National Hydrography Dataset (NHD; USGS, <http://nhd.usgs.gov/data.html>), and flow-direction layers from DEMs to delineate study streams. Riparian corridor boundaries were defined by 100-m buffers on either side of each stream that extended along the entire upstream length above the study-reach base.

Land cover for catchment and riparian-corridor polygons was quantified using annual data from the USDA Cropland Data Layer (2010–2012); (https://www.nass.usda.gov/Research_and_Science/Cropland/SARS1a.php) and four aggregate land-cover classes: (1) forest, (2) non-forest vegetation, (3) barren, and (4) developed. The “barren” class corresponds to bare soil or mine spoil areas, often indicating coal- or gas-related facilities. Because study years did not coincide with calendar years, we used the mean of annual results from 2010 and 2011, and from 2011 and 2012, to estimate land cover for each study year.

We used orthoimagery from the US Department of Agriculture, National Aerial Imagery Program, (2011–2012) and VBMP (2011) to determine coverage of VFs and ponds within a given catchment, both total and those that intersected stream channels (in-line). We used historical orthoimagery, communications with mining personnel, and normalized difference vegetation index (Sen et al., 2012) to estimate VF, pond, and stream construction dates.

2.3. Field and laboratory methods

2.3.1. Structural measures

Temperature data loggers (HOBO U-22; Onset Computer Corp., Bourne, MA) were deployed in the deepest pools of each study reach in July 2010 and recorded stream temperature at 30-min intervals through September 2012. Site visits were typically monthly but ranged from bi-weekly to bi-monthly. We measured dissolved oxygen, temperature, specific conductance (SC), and pH *in situ* with a multi-parameter meter (Hydrolab Quanta; Hach Instruments, Loveland, CO) during each site visit. We estimated discharge in a stable cross-section of each reach using the velocity-area technique when depth was sufficient to submerge the sensor (Marsh-McBirney Flo-Mate; Hach Instruments, Loveland, CO). When streams were flowing but too shallow for sensor submergence, we estimated discharge as half the lowest measured discharge at that stream. Using a concave spherical densiometer (Forestry Suppliers, Inc.; Jackson, MS), we estimated canopy cover seasonally between September 2010 and September 2011, and for each site visit from September 2011 through September 2012.

Water samples collected during each visit were filtered (0.45 μm , Durapore PVDF; EMD Millipore, Darmstadt, Germany) on site and transported to the lab on ice. Major cation and trace element samples were preserved in 1 + 1 HNO_3 prior to transport (APHA 2005). We used inductively coupled plasma-optical emis-

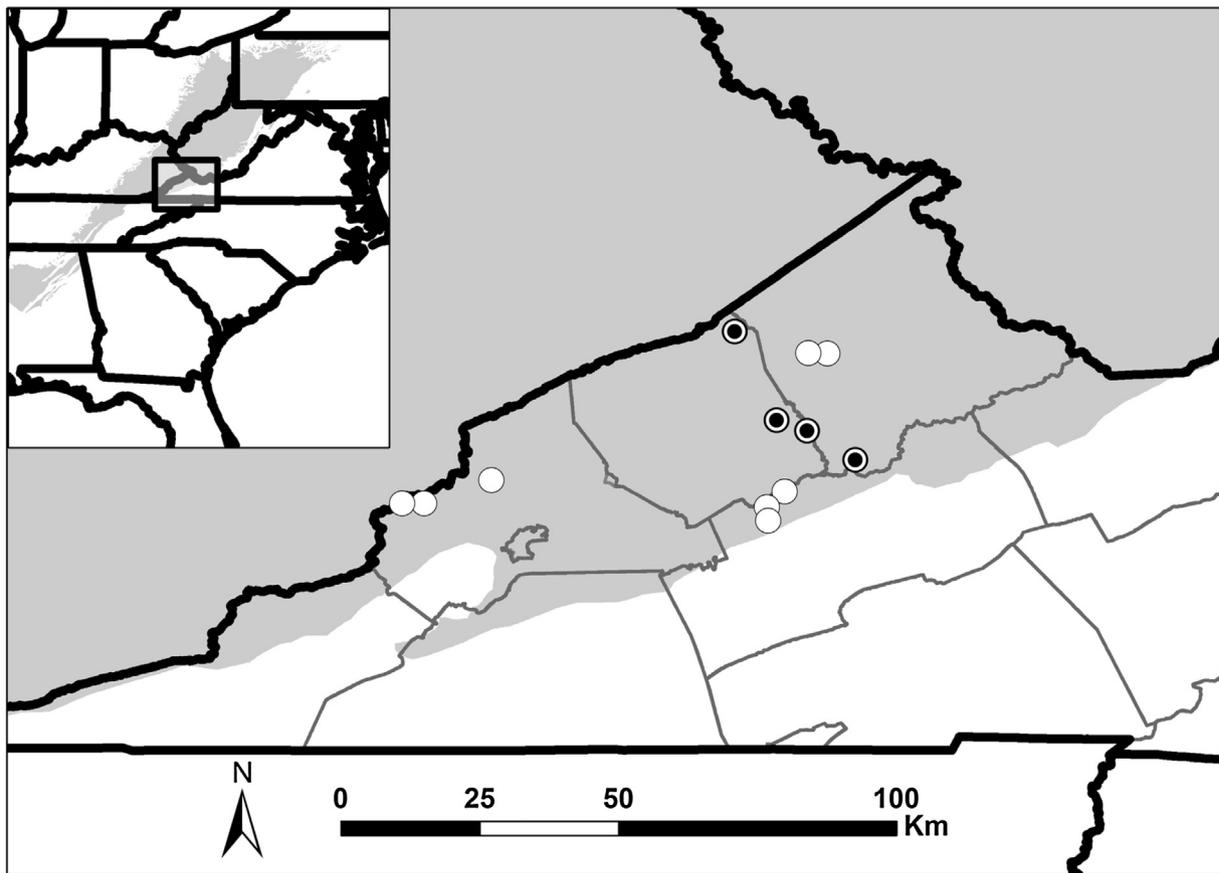


Fig. 1. Locations of study streams in southwestern Virginia, USA, with state and county boundaries. Reference streams are shown as symbols with black centers and constructed streams as white symbols. The inset shows study-area location within eastern USA. The Appalachian coalfield is denoted by gray background.

Table 1
Stream and catchment characterization.

Stream Code	Stream Order	Year Stream-reach Completed	Total Stream Length (m)	Stream Slope (%)	Mean Catchment Slope (%)	Catchment Area (ha)	Mean Catchment Aspect ^a (°)
Reference							
1. BIG	1st	n/a	3041	22	51	274	38
2. COP	1st	n/a	993	46	38	48	251
3. CRO	2nd	n/a	1585	10	44	211	228
4. MCB	1st	n/a	1265	31	52	79	260
Mean ± SE ^a	n/a	n/a	1721 ± 456 ^A	27 ± 8 ^A	46 ± 3.3 ^A	153 ± 54 ^A	n/a
Constructed							
5. GUE	2nd	2003 to 2004	563	45	51	144	201
6. CRI	2nd	1988 to 1989	953	16	37	266	84
7. CAL	1st	2006	1447	28	36	248	261
8. LLA	1st or 2nd	2005	2762	24	47	265	95
9. SCH	2nd	2006	2848	21	48	384	170
10. SEW	1st	2007 to 2008	115	25	27	34	219
11. SHO	1st	2008	157	35	31	26	188
12. STO	1st	2008	1331	15	43	92	3
Mean ± SE ^a	n/a	n/a	1272 ± 376 ^A	26 ± 4 ^A	40 ± 3.0 ^A	182 ± 45 ^A	n/a

^a For each numeric parameter, mean values connected by same upper-case letter are not significantly different ($\alpha = 0.05$).

sions spectrometry (Varian Vista MPX-CCD simultaneous ICP-OES, Varian Corp., Walnut Creek, CA) to determine dissolved concentrations of Ca^{2+} , Mg^{2+} , K^+ , Na^+ , Al , Cu , Fe , Mn , Se , and Zn . We measured SO_4^{2-} and Cl^- concentrations by ion chromatography (DX 500 IC; Dionex Corp., Sunnyvale, CA) and we estimated concentrations of HCO_3^- from titrimetrically determined alkalinity (Titralab 865; Radiometer Analytical, Lyon, France) and pH measurements (APHA 2005). Nitrogen as nitrate + nitrite ($\text{NO}_3^- + \text{NO}_2^- - \text{N}$) and as ammonium ($\text{NH}_4^+ - \text{N}$), and P as soluble reactive phosphate ($\text{PO}_4^{-3} - \text{P}$),

were determined using flow-injection colorimetry (Seal AutoAnalyzer 3, Seal Analytical, Mequon, WI).

We used the RBP single-habitat (i.e., riffle-run) approach (Barbour et al., 1999) to sample benthic macroinvertebrates at each stream in December 2010, May 2011, October 2011, and April 2012. We collected a two- m^2 sample composited from six 30-s kicks of $\sim 0.3 \text{ m}^2$ each using a D-frame kicknet (0.3-m wide, 500- μm mesh) in the uppermost 70 m of each 100-m study reach. Composited samples were preserved in 95% ethanol and processed to obtain randomized subsamples of 200 ($\pm 10\%$) individuals (VDEQ,

2008). We used keys (Merritt et al., 2008) to identify organisms to family-level or lower and entered these data into the Ecological Data Application System (EDAS v.3.0, 2000; Tetra Tech, Inc., Owings Mills, MD) to calculate taxonomic relative abundance, richness, and functional-feeding group metrics.

2.3.2. Functional measures

We identified a 60-m sub-reach extending upstream from the base of each 100-m study reach and bisected it into two 30-m segments for use in leaf breakdown (upstream) and litterfall (downstream) determinations.

We conducted litterfall sampling by securing ten direct-fall traps to stream banks, five equidistantly spaced on each side. Direct-fall traps were perforated 19-L buckets (627 cm² opening) lined with aluminum mesh cones. We composited litterfall at each stream during each site visit, returned it to the lab, removed any non-vegetative detritus, and sorted the OM into five fractions: wood, leaves/needles of woody taxa, reproductive parts of woody taxa (i.e., fruits, nuts, flowers), all herbaceous material, and unidentifiable detrital material. We dried samples at 65 °C for ≥ 5 d until a constant dry-mass (DM) was achieved.

We evaluated leaf breakdown in each upstream sub-reach by measuring mass lost from 6.5 g (± 0.01) of dry white oak (*Quercus alba*) leaves through serial collection of samples during 310- to 320-d deployments. Leaves were acquired from a single location, uniformly mixed, dried (65 °C for 48 h), weighed, and placed in bags of two mesh sizes: Coarse (1-cm) to allow access by macroinvertebrates, and fine (1-mm) to exclude macroinvertebrates (Chergui and Pattee 1990; O'Connor et al., 2000; Gessner and Chauvet 2002). During the first study year, 24 bags of each type were deployed in each stream in early December 2010, secured in pool-glides (transitions between pools and riffles) using paracord, and retrieved in triplicate monthly for three months and bimonthly thereafter. Procedures during the second study year were similar, except that 30 bags per stream were deployed in late November 2011.

Processing of leaves followed a modified version of methods presented by Benfield (2006). Leaf material was rinsed, then dried at 65 °C for ≥ 48 h until constant weight was achieved. Leaves were ground in a Wiley-Mill and sub-sampled. The ground sub-sample was weighed and ignited at 550 °C for 40 min, and residue was weighed to determine percentage of DM that was organic matter (% OM). Total DM of each leaf pack was then converted to ash-free DM (AFDM) using % OM values, and we determined percentage of AFDM remaining (% AFDM-r) for each pack relative to initial (0 d) values.

2.4. Data analysis

We determined litterfall input rates (g DM m² yr⁻¹) to each stream by summing areal inputs of each litterfall fraction from successive collections during each study year.

Leaf breakdown rates (k) in coarse and fine mesh were based on the first-order decay model (Olson 1963; Petersen and Cummins 1974; Webster and Benfield, 1986):

$$M_t = M_0 e^{-kt}$$

where,

M_0 = initial coarse or fine mesh leaf pack mass (%AFDM-r) at time 0,

M_t = coarse or fine mesh leaf pack mass (%AFDM-r) at time t ,

t = time, in days or degree-days, and

k = breakdown coefficient allowing (coarse), or excluding (fine) macroinvertebrates.

When inspection of % AFDM-r versus time plots indicated triplicate leaf-bag means had reached a lower limit, subsequent collections were excluded from breakdown calculations. Leaf bags used in breakdown calculations did not exceed 261 days of deployment. We derived breakdown rates and compared them on a stream-wise basis, treating each triplicate leaf pack on each stream-date combination as a replicate. We linearized data by natural-log transforming all % AFDM-r ($\ln[M_0, M_t]$) values, and specified days or degree-days as the covariate to determine breakdown rates in coarse- (k_{coarse}) and fine-mesh (k_{fine}) bags. Using an expanded, non-additive ANCOVA model with dummy coding (Kleinbaum et al., 1988), we tested the null hypothesis that each coefficient was equal to zero ($k=0$), and subsequently, we compared individual leaf breakdown rates for each constructed stream to the mean rate of reference streams using Bonferroni-adjusted simple contrasts (PROC GLM; SAS version 9.4, SAS Institute, Cary, NC).

We also calculated ratios of constructed-stream coarse-mesh breakdown coefficients to mean of reference coarse-mesh coefficients ($k_{\text{CONST}}: k_{\text{REF}}$) and the ratio of coarse- to fine-mesh breakdown coefficients ($k_{\text{coarse}}: k_{\text{fine}}$) to gain insight concerning the balance between macroinvertebrate and combined microbial + physicochemical processes mediating breakdown (Gessner and Chauvet, 2002). Differences between coarse- and fine-mesh leaf breakdown rates ($k_{\text{coarse}}-k_{\text{fine}}$) were also calculated as absolute indicators of breakdown attributable to benthic macroinvertebrate activity (Chergui and Pattee, 1990).

Variables summarizing the entire study period and variables measured only once were compared between stream types with unpaired t -tests, using transformations to satisfy residual assumptions if needed (Krenz, 2015). We substituted Mann-Whitney U tests and Welch's t -tests when iterative transformation did not resolve issues with non-normality or heteroscedacity of residuals, respectively. For variables that were constant among reference streams, we performed one-sample t -tests to evaluate whether constructed-stream means significantly deviated from zero following correction for the single, identical reference-stream level. We used Minitab version 17.1 (Minitab Inc., State College, PA) for all single-factor tests.

The two-factor (factor 1: constructed vs. reference, factor 2: year 1 vs. year 2) group-wise comparison component of this study is a proportionally replicated design (Zar, 1999). As such, we used parametric analyses when original or transformed data satisfied assumptions (Krenz, 2015). Means for variables measured across both years were tested for differences between stream types using a two-factor ANOVA general linear model procedure (Minitab version 17.1) with adjusted sums-of-squares to calculate F -statistics. Van Elteren's statistic (W'), with year as a blocking factor, was used for group-wise comparisons when transformation did not improve normality of residuals (PROC FREQ; SAS version 9.4, SAS Institute, Cary, NC).

To evaluate associations between riparian overstory and litterfall among constructed streams, we performed Pearson correlation analysis (JMP Pro 10; SAS Institute, Cary, NC) between riparian cover metrics and total litterfall, and as litterfall as leaves. Riparian land-cover and width scores were not significantly correlated with litterfall metrics and are not reported.

We performed Spearman rank correlation analysis (JMP Pro 10; SAS Institute, Cary, NC) within the constructed-stream group to determine any structural or litterfall variables that significantly ($\alpha = 0.05$) associated with total leaf breakdown rates (k_{coarse}). Cor-

Table 2
Mean (range) land cover and stream survey values, by stream type.

Structural Variable	Stream Type	
	Constructed	Reference ^a
Anthropogenic landscape features (%)		
In-line VF cover	3.59 (0.00–15.58)	0.00 (0.00–0.00)
In-line pond cover	0.13 (0.00–0.41)	0.00 (0.00–0.00)
Catchment VF cover	4.10 (0.00–15.58)	0.00 (0.00–0.00)
Catchment pond cover	0.23 (0.00–0.78)	0.03 (0.00–0.10)
Catchment VF+pond cover	4.33 (0.00–15.58)	0.03 (0.00–0.10)**
Catchment land cover (%)		
Forest	73.7 (41.2–95.7)	94.2 (83.7–99.9)*
Barren	9.9 (0.1–4.2)	0.1 (0–0.3)*
Developed	3.1 (0–6.2)	3.2 (0–8.4)
Non-forest vegetation	13.4 (4.2–27.7)	2.5 (0–7.8)*
Riparian land cover (%)		
Forest	76.78 (41.59–99.53)	93.93 (88.61–99.90)
Barren	1.28 (0.00–3.98)	0.02 (0.00–0.10)*
Developed	7.82 (0.00–23.26)	3.28 (0.00–8.47)
Non-forest vegetation	13.95 (0.47–40.56)	2.74 (0.00–6.75)
Stream survey		
Median sediment particle size (mm)	32 (2–128)	17 (8–32)
Total RBP habitat score	153 (140–167)	178 (175–182)***

^a Significant differences from constructed stream means are designated by * $p \leq 0.05$, ** $p \leq 0.005$, *** $p \leq 0.001$.

relations were analyzed independently for each year, and variables included in these analyses were either annual summaries (e.g., means) or were measured only once during the study.

Residual homoscedasticity and normality assumptions for all parametric models were tested using Levene and Ryan-Joiner diagnostics, respectively. Additionally, we visually evaluated normal quantile plots and histograms to ensure assumptions were met. Chemical analytes were excluded from means comparisons and correlation when $\geq 15\%$ of samples were below method detection limits (USEPA, 2006).

3. Results

3.1. Ecosystem structure

Mean coverage by VFs and ponds combined was significantly greater for constructed-stream catchments (4.3%) relative to reference catchments ($<0.1\%$; Table 2). When these anthropogenic features were considered separately, coverage by VFs and ponds did not significantly differ between site types on a catchment-wide basis or when considering only those features that intersected study streams. However, VFs were absent in all reference catchments and only a single small settling pond was present in one reference catchment near a gas well. Mean percent of catchment under forest cover was significantly greater for reference streams (94.2%) than for constructed streams (73.7%), and mean barren land cover was greater for constructed streams at catchment (9.9%) and riparian (1.3%) scales than for reference streams (catchment = 0.1%, riparian = 0.02%).

The RBP habitat scores were greater in reference streams than in constructed streams, and median particle size (D_{50}) did not differ significantly between stream types (Table 2).

Mean measured discharge (Q) and area-corrected discharge (Q/catchment area) did not differ significantly between stream types (Table 3). Mean canopy cover was less in constructed streams (22%) than in reference streams (75%), whereas constructed-stream SC ($904 \mu\text{S cm}^{-1}$) and pH (7.84) means were consistently elevated relative to reference means (SC = $85 \mu\text{S cm}^{-1}$; pH = 7.40). Dissolved SO_4^{2-} , HCO_3^- , $\text{NO}_3 + \text{NO}_2\text{-N}$, Ca^{2+} , Mg^{2+} , K^+ , and Na^+ were also elevated in constructed streams relative to reference means, but

Table 3
Physicochemical, canopy, water chemistry, and benthic macroinvertebrate means (\pm SE) by stream type.

Structural Variable	Stream Type	
	Constructed	Reference ^a
Periodic <i>in situ</i> metrics		
Specific conductance ($\mu\text{S cm}^{-1}$)	904 \pm 199	85 \pm 19***
pH	7.84 \pm 0.1	7.40 \pm 0.02**
Dissolved oxygen (mg L^{-1})	9.0 \pm 0.1	8.7 \pm 0.1
Q (L s^{-1})	35 \pm 12	20 \pm 5
Q/catch. area ($\text{L s}^{-1} \text{ ha}^{-1}$)	0.17 \pm 0.05	0.15 \pm 0.02
% Canopy cover	22 \pm 5	75 \pm 1*
Water chemistry (dissolved; mg L^{-1})		
[SO_4^{2-}]	381 \pm 107	11 \pm 2***
[HCO_3^-]	98 \pm 17	19 \pm 6***
[Cl^-]	5.2 \pm 2.4	3.5 \pm 1.7
[$\text{NO}_3 + \text{NO}_2\text{-N}$]	2.0 \pm 0.8	0.4 \pm 0.1*
[Ca^{2+}]	117 \pm 30	8 \pm 3***
[K^+] ^b	6.29 \pm 1.16	1.61 \pm 0.05**
[Mg^{2+}]	62.4 \pm 19.0	3.1 \pm 0.6***
[Na^+]	38 \pm 13	4 \pm 1*
Taxonomic Richness ^b		
Ephemeroptera	1.3 \pm 0.4	4.4 \pm 0.2***
Plecoptera	3.2 \pm 0.5	5.3 \pm 0.3*
Trichoptera	2.4 \pm 0.3	4.4 \pm 0.2***
EPT	6.9 \pm 0.6	14.1 \pm 0.4***
Total	13.4 \pm 1.4	21.7 \pm 0.4***
Relative abundance of taxa ^b		
% Ephemeroptera	13 \pm 4	31 \pm 1**
% Plecoptera	19 \pm 6	21 \pm 2
% Trichoptera	29 \pm 2	19 \pm 3*
% EPT	61 \pm 6	71 \pm 1
% 2 dominant taxa	67 \pm 5	37 \pm 2**
Functional feeding groups ^b		
% Shredders	17 \pm 5	21 \pm 2
% Scrapers	7 \pm 2	14 \pm 1
% Predators	6 \pm 1	10 \pm 1
% Collector-filterers	34 \pm 7	17 \pm 3*
% Collector-gatherers	35 \pm 5	37 \pm 2

^a Significant differences from constructed stream means are designated by * $p \leq 0.05$, ** $p \leq 0.005$, *** $p \leq 0.001$.

^b Fall 2010 GUE macroinvertebrate sample with only 79 total individuals is included in analysis.

dissolved trace elements (Al, Cu, Fe, Mn, Se, and Zn), $\text{NH}_4^+\text{-N}$, and $\text{PO}_4^{3-}\text{-P}$ were routinely below detection ($>15\%$ of samples) in both stream types (data not shown).

Several water temperature metrics varied by study year and stream type (Fig. 2). Constructed streams were warmer (daily mean = 12.6°C) and less thermally stable (mean daily fluctuation = 2.9°C) than reference streams (daily mean = 11.8°C ; mean daily fluctuation = 1.9°C). Average daily mean and minimum daily-mean temperatures for each study year (Fig. 2A, B), as well as minimum and mean daily temperature fluctuations (Fig. 2C, D), were all significantly greater in constructed streams than in reference streams. The second study-year was warmer than the first, regardless of stream type, as indicated by higher average (1st year = 11.8°C ; 2nd year = 12.8°C) and minimum daily-mean (1st year = 1.2°C ; 2nd year = 2.7°C) temperatures.

Numerous differences between stream types were evident for benthic macroinvertebrate metrics (Table 3). Ephemeroptera (E), Plecoptera (P), and Trichoptera (T), combined EPT, and total richness at the family-level were all significantly greater in reference streams relative to constructed streams. Relative mayfly abundances were elevated in reference streams relative to constructed-stream levels. Constructed streams had significantly higher mean proportions of collector-filterers (34%) relative to reference streams (17%), and relative abundances of caddisflies were also higher in constructed streams.

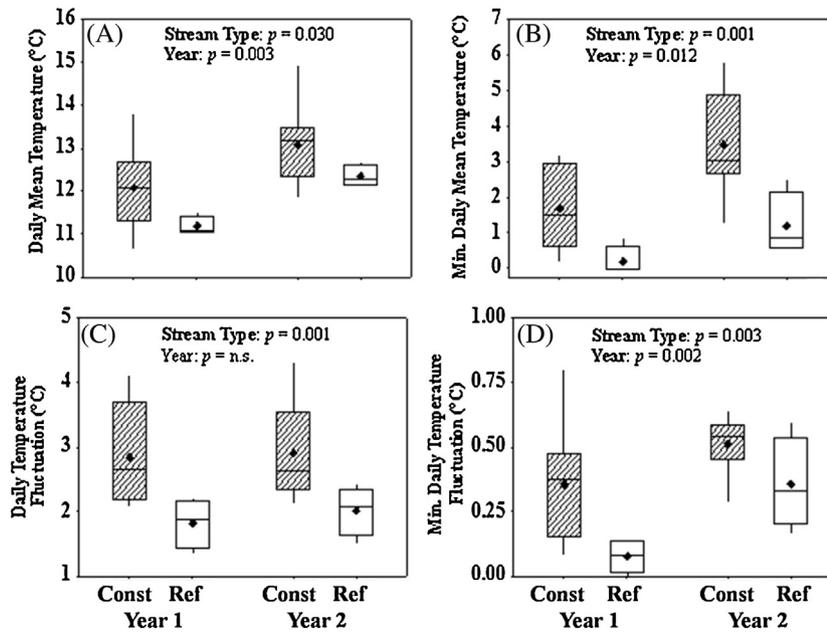


Fig. 2. Tests of stream-type and study-year main effects [*p*-value] on (A) daily mean temperature, (B) minimum daily mean temperature, (C) mean daily temperature fluctuation, and (D) minimum daily temperature fluctuation for constructed (Const) and reference (Ref) streams. Upper and lower box hinges represent the first and third quartiles, and whiskers indicate maximum and minimum observations within 1.5 times the upper and lower hinges. Medians and means are indicated by bisecting lines and closed diamonds within the boxes, respectively. Significant main effects for stream type and year are in bold. Interaction terms were not significant.

3.2. Litterfall

Total litterfall, and four measured litterfall component, inputs to reference streams exceeded corresponding inputs to constructed streams (Table 4). Mean reference-stream inputs of herbaceous OM, woody-plant leaves, and woody debris were roughly two, four, and nine times the corresponding constructed-stream means. Leaf inputs for one constructed stream with a developing canopy (SCH; 251 g DM m⁻² yr⁻¹) more closely approximated reference-stream levels (389 to 461 g DM m⁻² yr⁻¹), and relative dominance of leaf and woody inputs to SCH (73% and 13% of total litterfall, respectively) resembled the mean proportional dominance of leaves (75%) and wood (17%) in reference-stream litterfall. For each study year, mean canopy cover for constructed streams was positively correlated with total litterfall (1st year: $r=0.94$, $p=0.0006$; 2nd year: $r=0.86$, $p=0.0058$) and with litterfall as leaves (1st year: $r=0.94$, $p=0.0006$; 2nd year: $r=0.90$, $p=0.0027$).

3.3. Leaf breakdown

Although leaf breakdown rates varied considerably among constructed streams (Table 5), reference-stream leaf breakdown means significantly exceeded constructed-stream means regardless of unit (day⁻¹ or deg.-day⁻¹) or mesh type (Table 4). Differences among study years were detected for fine-mesh breakdown (day⁻¹ and deg.-day⁻¹) and coarse-mesh breakdown (deg.-day⁻¹); and a significant interaction ($p=0.043$) between stream type and study year was detected for coarse-mesh breakdown measured per degree day (Fig. 3). Tests of simple effects in presence of significant interaction revealed that during the second study year, mean coarse-mesh breakdown rates did not differ between constructed (2nd year k_{coarse} mean=0.0011 deg.-day⁻¹) and reference (2nd year k_{coarse} mean=0.0017 deg.-day⁻¹) streams after adjusting for temperature. Mean ratios of coarse-mesh breakdown rates in constructed streams relative to reference streams ($k_{\text{CONST}}: k_{\text{REF}}$) did not differ significantly between years (Table 4) regardless of units (day⁻¹ or deg.-day⁻¹).

Table 5

Leaf breakdown coefficients ($k \pm 95\%$ confidence limit) for individual streams and mean breakdown rates (\pm SE) by stream type for each study year.

Stream Code	k_{coarse} (day ⁻¹)		k_{fine} (day ⁻¹)	
	Year 1	Year 2	Year 1	Year 2
Constructed				
CAL	0.014 ± 0.007*	0.017 ± 0.004	0.007 ± 0.002*	0.003 ± 0.001*
CRI	0.010 ± 0.003*	0.010 ± 0.003*	0.008 ± 0.003*	0.003 ± 0.001*
GUE	0.007 ± 0.001*	0.007 ± 0.003*	0.005 ± 0.001*	0.004 ± 0.001*
LLA	0.013 ± 0.006*	0.013 ± 0.006	0.013 ± 0.005	0.008 ± 0.001
SCH	0.010 ± 0.002*	0.013 ± 0.005	0.006 ± 0.001*	0.010 ± 0.003*
SEW	0.005 ± 0.003*	0.004 ± 0.001*	0.004 ± 0.001*	0.003 ± 0.001*
SHO	0.019 ± 0.004	0.016 ± 0.008	0.016 ± 0.005	0.006 ± 0.001
STO	0.021 ± 0.008	0.019 ± 0.004	0.005 ± 0.001*	0.005 ± 0.001
Mean ± SE	0.012 ± 0.002	0.012 ± 0.002	0.008 ± 0.001	0.005 ± 0.001
Reference				
BIG	0.025 ± 0.006	0.019 ± 0.005	0.016 ± 0.004	0.006 ± 0.001
COP	0.024 ± 0.008	0.018 ± 0.004	0.018 ± 0.008	0.007 ± 0.002
CRO	0.024 ± 0.006	0.016 ± 0.003	0.011 ± 0.003	0.004 ± 0.001
MCB	0.021 ± 0.005	0.017 ± 0.005	0.012 ± 0.003	0.010 ± 0.003
Mean ± SE	0.024 ± 0.001	0.018 ± 0.001	0.014 ± 0.002	0.007 ± 0.001

* Individual constructed-stream breakdown rate was significantly different from reference-stream mean for that mesh-year combination; Bonferroni-adjusted α for 8 pair-wise comparisons each study year ($p \leq 0.00625$).

Within each stream type, mean coarse-mesh breakdown rates per day and per degree-day were significantly greater than corresponding fine-mesh means ($p \leq 0.005$). Coarse-to-fine-mesh ratios ($k_{\text{coarse}}: k_{\text{fine}}$) did not differ between stream types; however, significant stream-type effects were evident for differences between coarse- and fine-mesh breakdown rates ($k_{\text{coarse}}-k_{\text{fine}}$), blocking for year (Table 4). Although there was a significant main effect of study year on coarse-to-fine-mesh breakdown ratios when measured per day, this effect was not significant when testing rates per degree-day.

Some constructed streams functioned consistently at or near reference levels (Table 5). Specifically, SHO was not different from mean reference condition during either study year regardless of mesh type, and STO differed from the reference mean only for fine

Table 4
Litterfall, leaf breakdown rates, and breakdown indicator means (\pm SE) by stream type and study year.

Functional Variable	Stream Type ^a		Study Year ^a	
	Constructed	Reference	1st Year	2nd Year
Litterfall input (g DM m ⁻² yr ⁻¹)				
Detrital	6.7 \pm 1.7	12.6 \pm 2.5*	5.5 \pm 2.0	11.9 \pm 1.8
Herbaceous	24 \pm 6	13 \pm 2	13 \pm 3	27 \pm 8
Reproductive	6.4 \pm 1.5	18.0 \pm 4.4*	9.8 \pm 3.4	10.8 \pm 2.5
Wood	11 \pm 5	97 \pm 15***	41 \pm 16	39 \pm 13
Leaves ^b	94 \pm 19	420 \pm 11***	194 \pm 50	212 \pm 49
Total	142 \pm 26	561 \pm 22***	263 \pm 68	301 \pm 62
Leaf breakdown rates				
k_{coarse} (day ⁻¹)	0.012 \pm 0.001	0.021 \pm 0.001***	0.016 \pm 0.002	0.014 \pm 0.001
k_{coarse}^c (deg.-day ⁻¹)	0.0011 \pm 0.0001	0.0022 \pm 0.0002***	0.0016 \pm 0.0003	0.0013 \pm 0.0001*
k_{fine} (day ⁻¹)	0.007 \pm 0.001	0.011 \pm 0.002*	0.010 \pm 0.001	0.006 \pm 0.001**
k_{fine}^c (deg.-day ⁻¹)	0.0006 \pm 0.0001	0.0010 \pm 0.0002*	0.0009 \pm 0.0002	0.0005 \pm 0.0001*
Breakdown indicator metrics				
$k_{coarse} : k_{fine}$ (day ⁻¹ basis)	2.1 \pm 0.3	2.2 \pm 0.3	1.7 \pm 0.2	2.6 \pm 0.3*
$k_{coarse} : k_{fine}^c$ (deg.-day ⁻¹ basis)	2.3 \pm 0.4	2.4 \pm 0.3	1.9 \pm 0.3	2.7 \pm 0.4
$k_{CONST} : k_{REFmean}^d$ (coarse; day ⁻¹ basis)	0.6 \pm 0.1	–	0.5 \pm 0.1	0.7 \pm 0.1
$k_{CONST} : k_{REFmean}^d$ (coarse; deg.-day ⁻¹ basis)	0.5 \pm 0.1	–	0.4 \pm 0.1	0.7 \pm 0.1
$k_{coarse} - k_{fine}^b$ (day ⁻¹)	0.006 \pm 0.001	0.010 \pm 0.001*	0.006 \pm 0.001	0.008 \pm 0.001
$k_{coarse} - k_{fine}^b$ (deg.-day ⁻¹)	0.0005 \pm 0.0001	0.0011 \pm 0.0001*	0.0007 \pm 0.0002	0.0007 \pm 0.0001

^a Significant differences indicated for stream-type and study-year main effects are indicated by * $p \leq 0.05$, ** $p \leq 0.005$, *** $p \leq 0.001$.

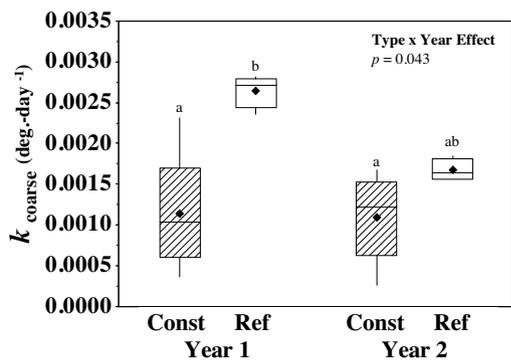


Fig. 3. Simple-effects tests (Tukey's HSD) of temperature-corrected leaf breakdown rates in coarse mesh in constructed (Const) and reference (Ref) streams. Boxes sharing a common letter are not significantly different. Significant interaction of stream type and study year is indicated in the upper right. For all other breakdown metrics, simple effects tests were not performed because interaction terms were not significant. See Table 4 for tests of main effects.

mesh during the first year. In contrast, three constructed streams (CRI, GUE, and SEW) had breakdown rates lower than reference mean for all mesh-size and study-year combinations.

Among constructed streams, coarse-mesh breakdown rates were negatively correlated with average daily mean temperatures for both years and with minimum daily mean temperature for the first year (Table 6). Catchment-level forest land cover and riparian forest cover were positively correlated with coarse-mesh leaf breakdown rates among constructed streams during the second year and during both years, respectively. Additionally, percent of catchment covered by all ponds and by ponds intersecting stream channels, were negatively associated with coarse-mesh breakdown for each year.

Daily mean temperatures of constructed streams correlated positively with catchment-level- (1st year: $\rho = 0.90$, $p = 0.002$; 2nd year: $\rho = 0.95$, $p < 0.001$) and in-line pond coverage (1st year: $\rho = 0.90$, $p = 0.002$; 2nd year: $\rho = 0.83$, $p = 0.011$), and negatively with riparian forest cover (1st year: $\rho = -0.76$, $p = 0.028$; 2nd year: $\rho = -0.81$, $p = 0.015$). No other non-temperature variables were

Table 6
Significant Spearman rank correlations (ρ) of coarse-mesh breakdown rates (k_{coarse}) with structural and litterfall variables for each study year among constructed streams^a.

Variable	k_{coarse} (day ⁻¹) ρ	
	1st Year (n = 8)	2nd Year (n = 8)
Catchment land cover (%)		
Forest	n.s.	0.76*
Non-forest vegetation	n.s.	-0.81*
Riparian land cover (%)		
Forest	0.71*	0.81*
Anthropogenic features (%)		
In-line pond cover	-0.78*	-0.71*
Catchment pond cover	-0.81*	-0.76*
Water temperature (°C)		
Average daily mean	-0.86**	-0.74*
Minimum daily mean	-0.74*	n.s.
Functional feeding groups		
% Collector-filterers	n.s.	-0.81*
Taxonomic relative abundance		
% Trichoptera	n.s.	-0.86**

^a Only correlations that were statistically significant for at least one study year-breakdown metric combination are presented, but all variables listed in Tables 2–4 were analyzed (n.s. = not significant).

* $p \leq 0.05$.

** $p \leq 0.01$.

consistently correlated with daily mean stream temperatures during both years. Coverage by the developed land-cover class, which indicates mining infrastructure and facilities in these catchments, was the only other variable that was positively associated with daily mean stream temperatures, but only during the first year ($\rho = 0.76$, $p = 0.028$).

Among benthic macroinvertebrate metrics, relative abundances of collector-filterer taxa and Trichoptera were negatively associated with coarse-mesh breakdown only during the second year (Table 6). Litterfall, RBP habitat, water chemistry, physical survey (e.g., D_{50}), and *in situ* water quality (i.e., SC, pH) measures were not significantly correlated with coarse-mesh breakdown rates among constructed streams during the study.

4. Discussion

4.1. Litterfall

Riparian litter inputs are a major source of allochthonous OM to forested headwater streams (Wallace et al., 1995b; Webster et al., 1995). Aquatic communities in forested streams typically depend on leaves as a basal energetic resource (e.g., Fisher and Likens 1973; Cummins, 1974; Vannote et al., 1980; Hall et al., 2000) and woody debris can affect physical structure (Sutherland et al., 2002; Valett et al., 2002), biogeochemical processes (Valett et al., 2002), and macroinvertebrate habitats (Palmer et al., 1996). For reference streams in this study, annual direct-fall of leaves, wood, and total litter, as well as the fraction of litterfall comprised of leaves (65% to 80%), were similar to values reported for other forested headwater streams in Appalachia (Webster et al., 1995; Wallace et al., 1995b; Benfield, 1997). In constructed streams, directly fallen wood, leaves, and total litter means were 25%, 22%, and 11% of reference-levels, respectively.

Riparian vegetation varied widely among constructed streams. Riparian areas of some constructed streams were characterized by planted trees, others by herbaceous forb plantings, and yet others by invasive non-native flora. Ontogeny of these riparian zones was not solely a factor of time elapsed since stream construction. The oldest constructed stream (CRI) had the lowest income of leaves (~3% of reference mean) and wood (~1% of reference mean) during the study. Conversely, subsidies of leaves (~60% of reference mean) and wood (~45% of reference mean) to SCH, a four-year-old stream initially, exceeded other constructed-stream inputs by at least twofold. Whereas the oldest constructed stream, CRI, was established without woody riparian plantings, planted non-native black alder (*Alnus glutinosa*) provided the majority of riparian subsidies to SCH. The remaining constructed streams were ≤ 7 years old at study onset and their riparian areas were dominated by herbaceous plantings and invasive flora.

Although quality of litter provided by late-successional native trees likely differs from that of black alder, SCH results demonstrate that relatively high levels of canopy cover (55% to 72%) are achievable in constructed-stream riparian areas and can provide fractional litter inputs that more closely resemble reference-stream quantities. In Appalachian streams impacted by clear-cut logging, successional processes returned litter inputs to reference-like quantities in as little as 5–10 years (Webster et al., 1988), although composition of such inputs can remain altered for 20 years or longer (Webster et al., 1990). In contrast, riparian successional processes did not produce developing forests during time-frames represented in this study. Other research has demonstrated that afforestation of reclaimed coal-mined lands in Appalachia is rarely rapid unless purposeful actions are taken to accelerate this process (e.g., Burger et al., 2005). Hill and Webster (1983) noted that OM contributions from upstream tributaries in one Appalachian river network were a major source of total OM in downstream reaches. Consequently, purposeful establishment of riparian canopy in constructed coal-field streams has implications for both on-site mitigation and downstream ecosystems.

Our riparian litterfall measures underestimate total OM income as we did not measure lateral or upstream inputs of litter or dissolved OM. However, in three steep-banked forested headwater streams of North Carolina an average of 90% of total OM inputs were as litterfall (Wallace et al., 1995b). Lateral inputs of particulate and dissolved OM fractions to constructed streams would likely be less than reference levels as most constructed stream banks had gently-sloping banks and were created from mine spoil. With little, if any, apparent use of topsoil. With less forest cover and more

barren lands than in reference-stream catchments, upstream OM inputs to constructed streams may also be diminished.

4.2. Leaf breakdown

We observed slower leaf breakdown in constructed streams on mined landscapes relative to forested reference streams, a pattern consistent with findings of Fritz et al. (2010) and Petty et al. (2013). In all three studies, mean reference-stream leaf breakdown rates in coarse mesh were approximately 1.5–2x faster than mean constructed-stream rates.

Benthic macroinvertebrates contributed to total leaf breakdown in both stream types as evidenced by coarse-mesh rates that significantly exceeded fine-mesh rates. However, relative to reference streams, leaf breakdown in constructed streams was inhibited as a direct result of both reduced macroinvertebrate-mediated processing (smaller $k_{\text{coarse}}-k_{\text{fine}}$ differences) and diminished contributions by microbial + physicochemical factors combined (lower k_{fine} rates).

Several structural factors differed between stream types and among constructed streams, and some of these are likely driving leaf-breakdown disparities. Indirect influences of stream temperature on leaf-breakdown processes have been widely recognized (Suberkropp et al., 1975; Paul et al., 1983; Webster and Benfield 1986; McArthur et al., 1988; Boulton and Boon 1991; Buzby and Perry, 2000; Royer and Minshall 2003; Robinson and Jolidon 2005; Fritz et al., 2010). Several studies have reported more rapid leaf breakdown in warmer streams (e.g., Suberkropp et al., 1975; Paul et al., 1983; Robinson and Jolidon 2005), and some authors (Buzby and Perry, 2000; Fritz et al., 2010) have predicted augmented microbial processing and macroinvertebrate maceration of leaves with increasing stream temperatures.

Conversely, several studies have documented leaf breakdown rates in cold streams that exceeded published values for warmer streams (Sedell et al., 1975; Short et al., 1980; Cowan et al., 1983); and Irons et al. (1994) suggested that leaf processing by microbes and macroinvertebrates may be positively and negatively related to stream temperature, respectively.

In concert with the findings of other authors (e.g., Rowe et al., 1996; Sponseller and Benfield, 2001; Fritz et al., 2010), we did not observe overriding positive effects of temperature on total (i.e., coarse-mesh) leaf breakdown. Rather, several lines of evidence support that leaf breakdown in cold streams can be relatively rapid in comparison with rates in warmer streams: (1) mean leaf breakdown (day^{-1} and $\text{deg.}\cdot\text{day}^{-1}$) was faster in the colder reference streams than in the warmer constructed streams; (2) mean reference-stream breakdown rates (day^{-1} and $\text{deg.}\cdot\text{day}^{-1}$) were nominally depressed in the warmer second year relative to the colder first year; and (3) among constructed streams, we observed a significant negative association between at least one temperature metric and total leaf breakdown during each year. Additionally, mean breakdown rates ($\text{deg.}\cdot\text{day}^{-1}$) were only similar between stream types during the warmer second year when reference-stream rates were depressed; and the two constructed streams with breakdown rates that did not differ from reference means during either study year (SHO and STO) had mean stream temperatures as low, or lower than, reference levels.

Catchment- and riparian-scale structural variables that were correlated with leaf breakdown were also consistently linked to stream temperature. Specifically, higher temperatures were significantly correlated with greater catchment-level and in-line pond coverage and with low levels of riparian forest cover. Other studies have documented warming effects of detention ponds (Herb et al., 2009; Jones and Hunt 2010) and riparian deforestation (Brown 1969; Lynch et al., 1984; Weatherley and Ormerod 1990) on streams, and negative relationships between temperature and

leaf breakdown can manifest indirectly via inhibitory temperature effects on benthic macroinvertebrates (e.g., Short et al., 1980; Cowan et al., 1983; Macdonald and Taylor, 2008; Taylor and Andrushchenko, 2014).

Despite evident thermal variation among streams, shredder relative abundances did not differ between stream types and were not significantly correlated with constructed-stream leaf breakdown rates, daily mean temperatures, pond cover metrics, or riparian forest cover. Life history and physiological responses of macroinvertebrates to warmer stream temperatures could potentially explain indirect, negative influences of temperature on leaf breakdown as well. For example, increasing temperatures have been shown to increase frequency of aquatic insect molts and respiratory stress (Camp et al., 2014), and molting periods have been associated with elevated insect mortality (Nebeker et al., 1996) as well as decreased feeding activity (Ayres and MacLean 1987; Stamp and Bowers, 1994; Rackauskas, 2006).

Evidence suggests that temperature is a major indirect driver of total breakdown differences between stream types and among constructed streams, but other factors may also have contributed to these differences. Although RBP habitat component scores were not significantly associated with breakdown rates among constructed streams (Krenz, 2015), several of these differed between constructed and forested stream types similarly to other studies (Northington et al., 2011; Petty et al., 2013). Additionally, other habitat-related phenomena also may have hampered breakdown of leaves in several constructed streams. Specifically, we observed evidence of anoxia (i.e., blackened leaves) driven by sedimentation in three constructed streams at some point during leaf deployments, and precipitate-coated leaves and streambed armoring in another. Oxygen-depletion is inhospitable to heterotrophic benthic microbes and macroinvertebrates, and hypoxia caused by sedimentation can reduce leaf processing (Cummins et al., 1980; Bunn 1988).

The one stream affected by mineral precipitates (CRI) had among the lowest mean relative abundance of shredders observed during either year (1st year=3%; 2nd year=0.3%). Additionally, its streambed was characterized by immobile cobbles and blocked interstices, and precipitate-coated leaves were common in leaf packs. Other authors have reported alteration of benthic habitats (e.g., Vuori, 1995) and macroinvertebrate assemblage structure (Woodcock and Huryn 2005; Bott et al., 2012; Macintosh and Griffiths, 2015) by streambed armoring. Moreover, coating of leaves by mineral precipitates has inhibited leaf breakdown processes in other mining-influenced streams (Gray and Ward 1983; Siefert and Mutz, 2001; Schlieff and Mutz, 2005).

A number of other structural factors differed between stream types and among constructed streams in this study, and indirect influences of many of these variables on leaf breakdown have been documented or implied by other studies. Nutrient enrichment stimulated leaf breakdown in some headwater streams of Appalachia (Gulis and Suberkropp 2003; Greenwood et al., 2007), and positive effects of discharge (Ferreira et al., 2006) and mechanical abrasion (Heard et al., 1999) on leaf breakdown have been reported in other lotic ecosystems. Measured discharge and median particle size of streambed substrates did not differ between stream types, and neither was significantly associated with constructed-stream breakdown rates. Therefore, it is unlikely that either of these factors were major drivers of the leaf-breakdown disparities that we observed. Constructed streams were enriched with $\text{NO}_3 + \text{NO}_2\text{-N}$ relative to reference levels. However, leaf breakdown in constructed streams was slower than in reference streams and $\text{NO}_3 + \text{NO}_2\text{-N}$ was not significantly associated with breakdown rates among constructed streams. This indicates that nutrient enrichment was not associated with breakdown-rate dissimilarities between stream types or among constructed streams.

Constructed and reference streams in this study differed greatly with respect to SC and, consequently, concentrations of major ions that are common in mining-influenced headwater streams (e.g. Ca^{++} , Mg^{++} , SO_4^{2-} , HCO_3^-). However SC and constituent ions did not correlate with leaf breakdown among constructed streams. These results are consistent with previous studies; as breakdown–SC relationships were not evident among the constructed streams studied by Petty et al. (2013). Additionally, we found that the two constructed streams with total leaf breakdown rates that did not differ from reference levels at any point during the study (SHO and STO) represented opposite ends of the constructed-stream SC gradient (mean SC values: 177 and 1789 $\mu\text{S cm}^{-1}$, respectively).

4.3. Functional relationships

Variation in natural leaf-accumulation quantities (Stout and Coburn, 1989), leaf quality (Melillo et al., 1982), or leaf types (Petersen and Cummins, 1974; Webster and Benfield, 1986) can alter breakdown rates, but we did not observe associations between litterfall components and constructed-stream breakdown rates. However, the inherent importance of an adequate supply of allochthonous OM to leaf-breakdown processes under ambient conditions can be illustrated by example. Although total rates of leaf breakdown in two constructed streams (i.e., STO, SHO) were consistently similar to reference means when leaf bags were artificially emplaced, leaf inputs did not exceed 26% of reference levels at either stream. Consequently, despite leaf breakdown potentials similar to reference, these constructed streams may not yield quantities or size fractions of OM similar to reference streams because of scarce source material. Conversely, total breakdown in SCH was slower than in STO or SHO, but rates were always within the top 50% of constructed streams and were coupled with the highest constructed-stream leaf inputs (~50–70% of any reference stream). These scenarios illustrate that replicating reference-level OM functions in constructed streams is contingent on providing sources of OM that mimic reference inputs.

4.4. Functional assessments

Our results can inform assessment efforts in coalfield streams. Gessner and Chauvet (2002) suggested that the ratio of leaf breakdown in coarse-mesh relative to fine-mesh ($k_{\text{coarse}}: k_{\text{fine}}$) may be a useful functional indicator of relative balance between macroinvertebrate- and microbe-mediated leaf processing. However, this metric did not discriminate between stream types in this study because of proportional reductions of breakdown rates in both mesh types among constructed streams. Moreover, if this ratio had been applied to our reference streams in the fashion suggested by those authors, only one of eight total observations would have been classified as “no clear evidence of impact”. In contrast, differences between coarse- and fine-mesh breakdown rates ($k_{\text{coarse}} - k_{\text{fine}}$; Chergui and Pattee, 1990) effectively discriminated between stream types. This indicates that an absolute indicator of leaf breakdown attributable to macroinvertebrates, as opposed to a proportional indicator, may be more appropriate for inclusion in future functional assessment protocols in mining-impacted constructed streams of Appalachia.

5. Conclusions

Allochthonous OM resources available to constructed streams in this study were depressed relative to reference levels. Additionally, total leaf breakdown rates were depressed in most constructed streams and resulted from reduced leaf processing by benthic macroinvertebrates and by combined microbial + physicochemical factors. We have identified reach-scale (stream temperature) and

higher-order structural variables (including riparian forest cover) that are associated with and appear to be controlling OM processes in constructed streams. Riparian forestation could potentially aid the functional restoration process in constructed streams by providing source material for OM functions and shade that moderates stream temperatures. Stream-mitigation practitioners can incorporate plantings of native riparian trees into stream designs and can manipulate soil-forming conditions during construction to promote successful establishment and growth of such plantings (Agouridis et al., 2010). Our results suggest that these activities could foster timelier establishment of reference-like OM processes in many constructed streams.

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References

- Agouridis, C.T., Wightman, S.J., Barton, C.D., Gumbert, A.A., 2010. *Planting a Riparian Buffer*. University of Kentucky Cooperative Extension Service Bulletin: ID-185. University of Kentucky, Lexington, KY.
- Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annu. Rev. Ecol. Syst.* 35, 257–284.
- Ayres, M., MacLean, S., 1987. Molt as a component of insect development: *Galerucella sagittariae* (Chrysomelidae) and *Epirrita autumnata* (Geometridae). *Oikos* 48, 273–279.
- Barbour, M.T., Gerritsen, J., Snyder, B.D., Stribling, J.B., 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish*, 2nd edition. U.S. Environmental Protection Agency, Office of Water, Washington, D.C (EPA 841-B-99-002).
- Benfield, E.F., 1997. Comparison of litterfall input to streams. *J. North Am. Benthol. Soc.* 16, 104–108.
- Benfield, E.F., 2006. Decomposition of leaf material. In: Hauer, F.R., Lamberti, G.A. (Eds.), *Methods in Stream Ecology*, 2nd edition. Academic Press, San Diego, CA, pp. 711–720.
- Bott, T.L., Jackson, J.K., McTammany, M.E., Newbold, J.D., Rier, S.T., Sweeney, B.W., Battle, J.M., 2012. Abandoned coal mine drainage and its remediation: impacts on stream ecosystem structure and function. *Ecol. Appl.* 22, 2144–2163.
- Boulton, A., Boon, P., 1991. A review of methodology used to measure leaf litter decomposition in lotic environments: time to turn over an old leaf? *Aust. J. Mar. Freshw. Res.* 42, 1–43.
- Brown, G.W., 1969. Predicting temperatures of small streams. *Water Resour. Res.* 5, 68–75.
- Bunn, S.E., 1988. Processing of leaf litter in two northern jarrah forest streams, western Australia: II. The role of macroinvertebrates and the influence of soluble polyphenols and inorganic sediment. *Hydrobiologia* 162, 211–223.
- Burger, J.A., Graves, D., Angel, P., Davis, V., Zipper, C.E., 2005. *The Forestry Reclamation Approach*. Forestry Reclamation Advisory, No. 2. Appalachian Regional Reforestation Initiative (ARRI). U.S. Department of the Interior, Office of Surface Mining, Pittsburgh, PA <http://arri.osmre.gov/>.
- Buzby, K.M., Perry, S.a., 2000. Modeling the potential effects of climate change on leaf pack processing in central Appalachian streams. *Can. J. Fish. Aquat. Sci.* 57, 1773–1783.
- Camp, A.A., Funk, D.H., Buchwalter, D.B., 2014. A stressful shortness of breath: molting disrupts breathing in the mayfly *Cloeon dipterum*. *Freshw. Sci.* 33, 695–699.
- Chergui, H., Pattee, E., 1990. The processing of leaves of trees and aquatic macrophytes in the network of the River Rhone. *Internationale Revue der gesamten Hydrobiologie und Hydrographie* 75, 281–302.
- Cowan, C.A., Oswood, M.W., Buttimore, C.A., Flanagan, P.W., Cowan, C.A., Oswood, M.W., Buttimore, C.A., Flanagan, P.W., 1983. Processing and macroinvertebrate colonization of detritus in an Alaskan subarctic stream. *Holarct. Ecol.* 6, 340–348.
- Cummins, K.W., Spengler, G.L., Ward, G.M., Speaker, R.M., Ovink, R.W., Mahan, D.C., Mattingly, R.L., 1980. Processing of confined and naturally retained leaf litter in a woodland stream ecosystem. *Limnol. Oceanogr.* 25, 952–957.
- Cummins, K.W., 1974. Structure and function of stream ecosystems. *Bioscience* 24, 631–641.
- Ferreira, V., Graça, M.A.S., de Lima, J.L.M.P., Gomes, R., 2006. Role of physical fragmentation and invertebrate activity in the breakdown rate of leaves. *Archiv für Hydrobiologie* 165, 493–513.
- Fisher, S.G., Likens, G.E., 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. *Ecol. Monogr.* 43, 421–439.
- Freeman, M.C., Pringle, C.M., Jackson, C.R., 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *J. Am. Water Resour. Assoc.* 43, 5–14.
- Fritz, K.M., Johnson, B.R., Walters, D.M., 2006. *Field Operations Manual for Assessing the Hydrologic Permanence and Ecological Condition of Headwater Streams*. U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C (EPA/600/R-06/126).
- Fritz, K.M., Fulton, S., Johnson, B.R., Barton, C.D., Jack, J.D., Word, D.A., Burke, R.A., 2010. Structural and functional characteristics of natural and constructed channels draining a reclaimed mountaintop removal and valley fill coal mine. *J. North Am. Benthol. Soc.* 29, 673–689.
- Futrell, M., Sforza, P., 2004. Converting the VBMP digital terrain model to a digital elevation model for raster analysis applications. In: McGee, J. (Ed.), *The Virginia Geospatial Newsletter*. Virginia Polytechnic Institute and State University, Blacksburg, VA, pp. 9–10.
- Gessner, M.O., Chauvet, E., 2002. A case for using litter breakdown to assess functional stream integrity. *Ecol. Appl.* 12, 498–510.
- Gray, L.J., Ward, J.V., 1983. Leaf litter breakdown in streams receiving treated and untreated metal mine drainage. *Environ. Int.* 9, 135–138.
- Greenwood, J.L., Rosemond, A.D., Wallace, J.B., Cross, W.F., Weyers, H.S., 2007. Nutrients stimulate leaf breakdown rates and detritivore biomass: bottom-up effects via heterotrophic pathways. *Oecologia* 151, 637–649.
- Gulis, V., Suberkropp, K., 2003. Leaf litter decomposition and microbial activity in nutrient-enriched and unaltered reaches of a headwater stream. *Freshw. Biol.* 48, 123–134.
- Hall, R.O., Wallace, J.B., Eggert, S.L., 2000. Organic matter flow in stream food webs with reduced detrital resource base. *Ecology* 81, 3445–3463.
- Harman, W., Starr, R., Carter, M., Tweedy, K., Clemmons, M., Suggs, K., Miller, C., 2012. *A Function-based Framework for Stream Assessment and Restoration Projects*. US Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, D.C (EPA 843-K-12-006).
- Heard, S.B., Schultz, G.A., Ogden, C.B., Griesel, T.C., 1999. Mechanical abrasion and organic matter processing in an Iowa stream. *Hydrobiologia* 400, 179–186.
- Herb, W.R., Mohseni, O., Stefan, H.G., 2009. Simulation of temperature mitigation by a stormwater detention pond. *J. Am. Water Resour. Assoc.* 45, 1164–1178.
- Hill, B.H., Webster, J.R., 1983. Aquatic macrophyte contribution to the New River organic matter budget. In: Fontaine, T.D., Bartell, S.M. (Eds.), *Dynamics of Lotic Ecosystems*. Ann Arbor Science Publishing, Ann Arbor, MI, pp. 273–282.
- Hooke, R.L., 1999. Spatial distribution of human geomorphic activity in the United States: comparison with rivers. *Earth Surf. Processes Landforms* 24, 687–692.
- Irons, J.G., Oswood, M.W., Stout, R.J., Pringle, C.M., 1994. Latitudinal patterns in leaf litter breakdown: is temperature really important? *Freshw. Biol.* 32, 401–411.
- Iversen, T.M., Thorup, J., Skriver, J., 1982. Inputs and transformation of allochthonous particulate organic matter in a headwater stream. *Ecography* 5, 10–19.
- Jones, M.P., Hunt, W.F., 2010. Effect of storm-water wetlands and wet ponds on runoff temperature in trout sensitive waters. *J. Irrig. Drain. Eng.* 136, 656–661.
- Kleinbaum, D.G., Kupper, L.L., Muller, K.E., 1988. *Applied Regression Analysis and Other Multivariable Methods*, 2nd edition. PWS-Kent Publishing Company, Boston, MA.
- Krenz, R.J., 2015. *Organic Matter Processes of Constructed Streams and Associated Riparian Areas in the Coalfields of Southwest Virginia*. Ph.D. Dissertation. Virginia Tech, Blacksburg.
- Leopold, L.B., Wolman, M.G., Miller, J.P., 1964. *Fluvial Processes in Geomorphology*. WH Freeman, San Francisco.
- Lynch, J.A., Rishel, G.B., Corbett, E.S., 1984. Thermal alteration of streams draining clearcut watersheds: quantification and biological implications. *Hydrobiologia* 111, 161–169.
- Macdonald, E.E., Taylor, B.R., 2008. Factors influencing litter decomposition rates in upstream and downstream reaches of river systems of eastern Canada. *Archiv für Hydrobiologie* 172, 71–86.
- Macintosh, K.A., Griffiths, D., 2015. Changes in epilithic biomasses and invertebrate community structure over a deposit metal concentration gradient in upland headwater streams. *Hydrobiologia* 760, 159–169.
- McArthur, J., Barnes, J., Hansen, B., Leff, L., 1988. Seasonal dynamics of leaf litter breakdown in a Utah alpine stream. *J. North Am. Benthol. Soc.* 7, 44–50.
- McClain, M.E., Boyer, E.W., Dent, C.L., Gergel, S.E., Grimm, N.B., Groffman, P.M., Hart, S.C., Harvey, J.W., Johnston, C.A., Mayorga, E., McDowell, W.H., Pinay, G., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems* 6, 301–312.
- Melillo, J.M., Aber, J.D., Muratore, J.F., 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. *Ecology* 63, 621–626.
- Merritt, R.W., Cummins, K.W., Berg, M.B., 2008. *An Introduction to the Aquatic Insects of North America*. Kendall Hunt Publishing, Dubuque, IA.
- Meyer, J.L., Strayer, D.L., Wallace, J.B., Eggert, S.L., Helfman, G.S., Leonard, N.E., 2007. The contribution of headwater streams to biodiversity in river networks. *J. Am. Water Resour. Assoc.* 43, 86–103.

- Nebeker, A., Onjukka, S., Stevens, D., Chapman, G., 1996. Effect of low dissolved oxygen on aquatic insect life stages of the caddisfly *Clistoronia magnifica* (Limnephilidae). *Arch. Environ. Contam. Toxicol.* 31, 453–458.
- Northington, R.M., Benfield, E.F., Schoenholtz, S.H., Timpano, A.J., Webster, J.R., Zipper, C.E., 2011. An assessment of structural attributes and ecosystem function in restored Virginia coalfield streams. *Hydrobiologia* 671, 51–63.
- O'Connor, P., Covich, A., Scatena, F., Loope, L., 2000. Non-indigenous bamboo along headwater streams of the Luquillo Mountains, Puerto Rico: leaf fall, aquatic leaf decay and patterns of invasion. *J. Trop. Ecol.* 16, 499–516.
- Olson, J.S., 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* 44, 322–331.
- Omernik, J.M., 1987. Map supplement: ecoregions of the conterminous United States. *Ann. Assoc. Am. Geogr.* 77, 118–125.
- Palmer, M.A., Hondula, K.L., 2014. Restoration as mitigation: analysis of stream mitigation for coal mining impacts in southern Appalachia. *Environ. Sci. Technol.* 48, 10552–10560.
- Palmer, M.A., Arensburger, P., Martin, A.P., Denman, D., 1996. Disturbance and patch specific responses: the interactive effects of woody debris and floods in lotic invertebrates. *Oecologia* 105, 247–257.
- Palmer, M.A., Bernhardt, E.S., Schlesinger, W.H., Eshleman, K.N., Foufoula-Georgiou, E., Hendryx, M.S., Lemly, A. D., Likens, G.E., Loucks, O.L., Power, M.E., White, P.S., Wilcock, P.R., 2010. Mountaintop mining consequences. *Science* 327, 148–149.
- Paul Jr., R.W., Benfield, E.F., Cairns Jr., J., 1983. Dynamics of leaf processing in a medium-sized river. In: Fontaine, T.D. (Ed.), *Dynamics of Lotic Ecosystems*. Ann Arbor science publishing, Ann Arbor, MI, pp. 403–423.
- Petersen, R.C., Cummins, K.W., 1974. Leaf processing in a woodland stream. *Freshw. Biol.* 4, 343–368.
- Petty, J.T., Gingerich, G., Anderson, J.T., Ziemkiewicz, P.F., 2013. Ecological function of constructed perennial stream channels on reclaimed surface coal mines. *Hydrobiologia* 720, 39–53.
- Rackauskas, C., 2006. Molting inhibits feeding in a grasshopper. *J. Orthopt. Res.* 15, 187–190.
- Robinson, C.T., Jolidon, C., 2005. Leaf breakdown and the ecosystem functioning of alpine streams. *J. North Am. Benthol. Soc.* 24, 495.
- Rowe, J.M., Perry, W.B., Perry, S.A., 1996. Leaf litter processing in West Virginia mountain streams: effects of temperature and stream chemistry. In: Hom, J., Birdsey, R., O'Brian, K. (Eds.), *Proceedings, 1995 Meeting of the Northern Global Change Program; General Technical Report NE-214*. U.S. Department of Agriculture, Forest Service (USFS), Northeastern Forest Experiment Station, Radnor, PA.
- Royer, T.V., Minshall, G.W., 2003. Controls on leaf processing in streams from spatial-scaling and hierarchical perspectives. *J. North Am. Benthol. Soc.* 22, 352–358.
- Schlieff, J., Mutz, M., 2005. Long-term leaf litter decomposition and associated microbial processes in extremely acidic (pH <3) mining waters. *Archiv für Hydrobiologie* 164, 53–68.
- Sedell, J.R., Triska, F.J., Triska, N.S., 1975. The processing of conifer and hardwood leaves in two coniferous forest streams: i: Weight loss and associated invertebrates. *Verhandlung Internationale Vereinigung Limnologie* 19, 1617–1627.
- Sen, S., Zipper, C.E., Wynne, R.H., Donovan, P.F., 2012. Identifying revegetated mines as disturbance/recovery trajectories using an interannual landsat chronosequence. *Photogramm. Eng. Remote Sens.* 78, 223–235.
- Short, R., Canton, A.S.P., Ward, J.V., 1980. Detrital processing and associated macroinvertebrates in a Colorado mountain stream. *Ecology* 61, 727–732.
- Siefert, J., Mutz, M., 2001. Processing of leaf litter in acid waters of the post-mining landscape in Lusatia, Germany. *Ecol. Eng.* 17, 297–306.
- Sparks, J., Townsend, J., Hagman, T., Messer, D., 2003. Stream Assessment Protocol for Headwater Streams in the Eastern Kentucky Coalfield Region. *Aquatic Resources News, a Regulatory Newsletter*. United States Army Corps of Engineers (USACE), Louisville District, Louisville, KY.
- Sponseller, R.A., Benfield, E.F., 2001. Influences of land use on leaf breakdown in southern Appalachian headwater streams: a multiple-scale analysis. *J. North Am. Benthol. Soc.* 20, 44–59.
- Stamp, N., Bowers, M., 1994. Effect of temperature and leaf age on growth versus moulting time of a generalist caterpillar fed plantain (*Plantago lanceolata*). *Ecol. Entomol.* 19, 199–206.
- Stout, B.M., Coburn, C.B., 1989. Impact of highway construction on leaf processing in aquatic habitats of eastern Tennessee. *Hydrobiologia* 178, 233–242.
- Suberkropp, K., Klug, M.J., Cummins, K.W., 1975. Community processing of leaf litter in woodland streams. *Verhandlung Internationale Vereinigung Limnologie* 19, 1653–1658.
- Sutherland, a.B., Meyer, J.L., Gardiner, E.P., 2002. Effects of land cover on sediment regime and fish assemblage structure in four southern Appalachian streams. *Freshw. Biol.* 47, 1791–1805.
- Taylor, B.R., Andrushchenko, I.V., 2014. Interaction of water temperature and shredders on leaf litter breakdown: a comparison of streams in Canada and Norway. *Hydrobiologia* 721, 77–88.
- Townsend, P.A., Helmers, D.P., Kingdon, C.C., McNeil, B.E., de Beurs, K.M., Eshleman, K.N., 2009. Changes in the extent of surface mining and reclamation in the Central Appalachians detected using a 1976–2006 Landsat time series. *Remote Sens. Environ.* 113, 62–72.
- USDA-NRCS, 2007. Stream Restoration Design, National Engineering Handbook Part 654. U.S. Department of Agriculture, Natural Resources Conservation Service, Washington, D.C (H-210-NEH-654).
- EPA, 2005. Mountaintop Mining/Valley Fills in Appalachia Final Programmatic Environmental Impact Statement. U.S. Environmental Protection Agency, Region 3 Office, Philadelphia, PA (EPA 9-03-R-05002).
- EPA, 2006. Data Quality Assessment: Statistical Methods for Practitioners EPA QA/G-9S. US Environmental Protection Agency, Office of Environmental Information, Washington, D.C (EPA/240/B-06/003).
- EPA, 2011. The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields. U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, Washington, D.C (EPA/600/R-09/138F).
- VDEQ, 2008. Biological Monitoring Program Quality Assurance Project Plan for Wadeable Streams and Rivers. Virginia Department of Environmental Quality, Richmond, VA.
- Valet, H.M., Crenshaw, C.L., Wagner, P.F., 2002. Stream nutrient uptake, forest succession, and biogeochemical theory. *Ecology* 83, 2888–2901.
- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. *Can. J. Fish. Aquat. Sci.* 37, 130–137.
- Vidon, P., Allan, C., Burns, D., Duval, T.P., Gurwick, N., Inamdar, S., Lowrance, R., Okay, J., Scott, D., Sebestyen, S., 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. *J. Am. Water Resour. Assoc.* 46, 278–298.
- Vuori, K.-M., 1995. Direct and indirect effects of iron on river ecosystems. *Ann. Zool. Fenn.* 32, 317–329.
- Wallace, J.B., Webster, J.R., Meyer, J.L., 1995a. Influence of log additions on physical and biotic characteristics of a mountain stream. *Can. J. Fish. Aquat. Sci.* 52, 2120–2137.
- Wallace, J.B., Whiles, M.R., Eggert, S., Cuffney, T.F., Lugthart, G.J., Chung, K., 1995b. Long-term dynamics of coarse particulate organic matter in three Appalachian mountain streams. *J. North Am. Benthol. Soc.* 14, 217–232.
- Weatherley, N.S., Ormerod, S.J., 1990. Forests and the temperature of upland streams in Wales: a modelling exploration of the biological effects. *Freshw. Biol.* 24, 109–122.
- Webster, J.R., Benfield, E.F., 1986. Vascular plant breakdown in freshwater ecosystems. *Annu. Rev. Ecol. Syst.* 17, 567–594.
- Webster, J.R., Benfield, E.F., Golladay, S.W., Kazmierczak Jr., R.F., Perry, W.B., Peters, G.T., 1988. Effects of watershed disturbance on stream seston characteristics. In: Swank, W.T., Crossley, D.A. (Eds.), *Forest Hydrology and Ecology at Coweeta*. Springer, New York, NY, pp. 279–294.
- Webster, J.R., Golladay, S.W., Benfield, E.F., D'Angelo, D.J., Peters, G.T., 1990. Effects of forest disturbance on particulate organic matter budgets of small streams. *J. North Am. Benthol. Soc.* 9, 120–140.
- Webster, J.R., Covich, A.P., Tank, J.L., Crockett, T.V., 1994. Retention of coarse organic particles in streams in the southern Appalachian mountains. *J. North Am. Benthol. Soc.* 13, 140–150.
- Webster, J.R., Wallace, J.B., Benfield, E.F., 1995. Organic processes in streams of the eastern United States. In: Cushing, C.E., Cummins, K.W., Minshall, G.W. (Eds.), *Ecosystems of the World: River and Stream Ecosystems*. Elsevier, New York, NY, pp. 117–187.
- Webster, J.R., Tank, J.L., Wallace, J.B., Meyer, J.L., Eggert, S.L., Ehrman, T.P., Ward, B.R., Bennett, B.L., Wagner, P.F., Mctammany, M.E., 2000. Effect of litter exclusion and wood removal on phosphorus and nitrogen retention in a forest stream. *Verhandlungen des Internationalen Verein Limnologie* 27, 1337–1340.
- Woodcock, T.S., Huryn, A.D., 2005. Leaf litter processing and invertebrate assemblages along a pollution gradient on a Maine (USA) headwater stream. *Environ. Pollut.* 134, 363–375.
- Zar, J.H., 1999. *Biostatistical Analysis*, 4th edition. Prentice-Hall, Inc., Upper Saddle River, NJ.