J. N. Am. Benthol. Soc., 2010, 29(2):673–689

DOI: 10.1899/09-060.1
Published online: 6 April 2010

Structural and functional characteristics of natural and constructed channels draining a reclaimed mountaintop removal and valley fill coal mine

Ken M. Fritz[1,6], **Stephanie Fulton**[2,7], **Brent R. Johnson**[1,8], **Chris D. Barton**[3,9], **Jeff D. Jack**[4,10], **David A. Word**[4,11], AND **Roger A. Burke**[5,12]

[1] *Office of Research and Development, National Exposure Research Laboratory, US Environmental Protection Agency, Cincinnati, Ohio 45268 USA*
[2] *Region 4, Watershed Protection Division, US Environmental Protection Agency, Atlanta, Georgia 30303 USA*
[3] *Department of Forestry, University of Kentucky, Lexington, Kentucky 40546 USA*
[4] *Department of Biology, University of Louisville, Louisville, Kentucky 40292 USA*
[5] *Office of Research and Development, National Exposure Research Laboratory, US Environmental Protection Agency, Athens, Georgia 30605 USA*

Abstract. Mountaintop removal and valley fill (MTR/VF) coal mining has altered the landscape of the Central Appalachian region in the USA. Among the changes are large-scale topographic recontouring, burial of headwater streams, and degradation of downstream water quality. The goals of our study were to: 1) compare the structure and function of natural and constructed stream channels in forested and MTR/VF catchments across ephemeral, intermittent, and perennial flow regimes and 2) assess the relationship between leaf litter breakdown and structural measures, such as the habitat assessments currently used by regulatory agencies. Specific conductance of stream water was, on average, 36 to 57× higher at perennial reaches below valley fills than at perennial reaches in forested catchments, whereas pH was circumneutral in both catchment types. Channel habitat and invertebrate assemblages in litter bags differed between forested streams and constructed channels in VF catchments. Invertebrate density, diversity, and biomass were typically higher in litter bags from forested catchments than from VF catchments. No differences in fungal biomass, estimated as ergosterol concentration, were detected between litter bags from forested and VF catchments. Breakdown of oak (*Quercus alba*) leaves was slower at perennial and intermittent reaches in VF catchments than at perennial and intermittent reaches in forested catchments. However, breakdown rates did not differ between ephemeral reaches on VFs and in forested catchments. Breakdown rates of oak leaves were significantly correlated to conductivity at perennial and intermittent reaches and to shredder diversity across all reaches, but were not correlated with habitat assessment scores currently being used to determine compensatory mitigation. Landuse changes associated with MTR/VF have detrimental consequences to headwater stream function that are not adequately evaluated using the prevalent habitat assessment.

Key words: mountaintop removal, valley fill, coal mining, litter breakdown, organic matter processing, hydrologic permanence, reclaimed mine, rapid functional methods, stream assessment, mitigation.

Worldwide coal production has increased over the last 30 y from 4.2 billion to 7 billion tons/y to meet increasing global energy demands (USDOE-EIA 2008c). In the USA, where coal-fired power plants supply half of the electricity, coal consumption for electricity is expected to increase 42% from 2008 to 2030 (USDOE-EIA 2008b) despite changing energy

[6] E-mail addresses: fritz.ken@epa.gov
[7] fulton.stephanie@epa.gov
[8] johnson.brent@epa.gov
[9] barton@uky.edu
[10] Deceased
[11] daword@gmail.com
[12] burke.roger@epa.gov

policies intended to encourage more use of alternative fuels. Furthermore, policies and technologies aimed to reduce CO_2 emissions from coal-fired power plants (e.g., Anderson and Newell 2004) do not address the direct water-quality problems associated with large-scale coal extraction. In 2007, the Central Appalachian Mountains produced nearly 20% of US coal, and this region alone ranks 7th internationally in coal production (USDOE-EIA 2008a, c).

Historically, the mountainous terrain of the Central Appalachians limited surface mining to outcrops of coal seams along topographic contours (i.e., auger and contour or highwall mining). However, a relatively recent method of coal mining called mountaintop removal and valley fill (MTR/VF) has contributed to a dramatic increase in coal production from surface mining in southern West Virginia and eastern Kentucky (Greene and Raney 1979, Robins 1979, Slonecker and Benger 2002). For example, coal production from eastern Kentucky surface mines increased > 400% from 1969 to 1998 (Kentucky Geological Survey 2009). The MTR/VF method has existed since the 1960s, but a combination of incentives for coal production, amendments to the US Clean Air Act to reduce sulfur emissions, and technological advances have made MTR/VF economically feasible (Fox 1999, Duffy 2003, Szwilski et al. 2001).

MTR/VF begins with removal (through blasting and excavation) of vegetation, soil, and layers of sedimentary rock (i.e., overburden or spoil) above underlying coal seams, and then the coal is extracted. The overburden is deposited into adjacent valleys creating hollow or valley fills (VFs). To meet performance standards set under the Surface Mining Control and Reclamation Act of 1977 (SMCRA; implemented and enforced by the Office of Surface Mining [OSM] in the US Department of Interior, except where federally delegated to qualified states), reclaimed mines must be physically stable. In addition to compaction and terracing, SMCRA performance standards require that VFs have permanent diversion channels (i.e., groin drains) to contain and direct runoff, minimize erosion, and prevent fill destabilization (Appendix 1A; available online from: http://dx.doi.org/10.1899/09-060.1.s1). Depending upon design and state regulations, VFs have a rock underdrain, and either a central drain or 2 perimeter groin drains that join at the base or toe of the fill. These constructed channels are required to contain floods associated with at least 100-y, 24-h precipitation events and to minimize contribution of suspended solids to downstream waters. The bed and banks of these channels must be constructed with durable, nonacid-, nontoxin-forming boulders (>1 m diameter; Appendix 1B; available online from: http://dx.doi.org/10.1899/09-060.1.s1).

In an assessment of VFs (>12 ha) in Kentucky, the US Environmental Protection Agency found that between 1985 and 1999, >660 km of headwater streams were permanently buried (USEPA 2005). Filling of waters of the US is regulated under Section 404 of the US Clean Water Act (CWA), and permitting of such activities is authorized by the US Army Corps of Engineers (USACE) and state water-quality agencies (under CWA §401). Under CWA Section 404(b)(1), regulatory agencies must consider potential impacts to the stream functions and values when determining compensatory mitigation requirements for aquatic resources unavoidably lost or adversely affected by authorized activities. The USACE did not typically require compensatory mitigation for coal mining activities prior to the authorization of the 2002 Nationwide Permits (NWP). Since 2002, compensatory mitigation has been required for all coal mining activities authorized under Section 404, including NWPs (i.e., NWP 21, Surface Coal Mining Operations; NWP 49, Coal Remining Activities, and NWP 50, Underground Coal Mining Activities). Since 2004, on-site mitigation credit has been given to coal mine operators for constructed drains. In the past, the linear distance of streams impacted by mining in the Central Appalachians was the only factor used in estimating the mitigation necessary for buried or impacted streams. In an attempt to incorporate the hydrologic class (i.e., ephemeral, intermittent, and perennial) and stream quality affected by a permitted activity, the Louisville Army Corps District developed a method (Sparks et al. 2003a, b) that combines specific conductance of water, the USEPA rapid bioassessment protocol (RBP) habitat assessment score (Barbour et al. 1999), and, where possible, a macroinvertebrate bioassessment index score (Pond and McMurray 2002). The Huntington Army Corps District also has used hydrologic permanence and the RBP habitat assessment score for determining mitigation requirements. A recent court case (*Ohio Valley Environmental Coalition v. USACE*, Southern District West Virginia 2007) has questioned whether such assessments used by regulatory agencies have adequately characterized the stream functions that will be lost when assessing fill permits and compensatory mitigation associated with MTR/VF activities.

Stream structure refers to the pattern or organization of features within a system (e.g., bed particle size distribution, macroinvertebrate diversity), whereas stream functions are the processes and rates of a system (e.g., metabolism, nutrient uptake; Bunn and Davies 2000). It usually is assumed that structure and function are closely related (e.g., Schindler 1987, Wallace et al. 1996), but this relationship is poorly understood. Structural measures have been used to a greater extent than functional measures to characterize the integrity of aquatic systems. However, functional measures have been advocated for stream assessments in the past (Matthews et al. 1982), and recent suggestions have been made to use them for regulatory purposes (Meyer 1997, Gessner and Chauvet 2002, Davies and Jackson 2006). Combined use of structural and functional measures would support regulatory agencies in meeting their regulatory obligations under the Clean Water Act, might provide better information regarding the integrity of water bodies, and would help integrate our understanding across levels of organization (e.g., population, species, ecosystem; Bunn and Davies 2000).

Assessments of the coal mining impacts on streams have been focused mainly on the effects and remediation of acid mine drainage (AMD; Kelly 1988) rather than on the effects of filling headwaters. Aside from permanent burial of upstream reaches, several patterns have been consistently documented from the relatively few studies on the impacts of MTR/VF on streams.

These patterns include elevated total dissolved solids (TDS) or specific conductance and specific ions (e.g., SO_4^{2-}, Mg^{2+}, Ca^{2+}, HCO_3^-) in stream water, stabilized water temperature, elevated base flow, increased flood risk, and impaired macroinvertebrate communities, particularly reductions in the abundance and diversity of Ephemeroptera (Wiley et al. 2001, Messinger and Paybins 2003, Phillips 2004, Hartman et al. 2005, USEPA 2005, Pond et al. 2008, Simmons et al. 2008). As in other forms of mining, elevated TDS downstream of VFs is a result of exposing fractured rock to weathering and percolation. Coal mine operators have received on-site mitigation credit for constructed channels, but we are not aware of any published studies that compared the structure and function of such constructed channels to natural channels.

Our objectives were to: 1) compare habitat and biological characteristics (structure) and litter breakdown (function) in natural and constructed channels in forested and MTR/VF catchments across a hydrological permanence gradient and 2) assess the relationship between litter breakdown and structural measures. RBP habitat assessment was one of our structural measures because it has been used to assess the potential impacts to the stream structure and function when determining compensatory mitigation requirements for authorized burial of headwater streams. We used litter breakdown as our functional measure because it is an important function of forested headwater streams in the Appalachians, and previous work endorses its use as a functional indicator (Gessner and Chauvet 2002, Young et al. 2008). We predicted that litter breakdown would differ between mined and forested catchments based on documented impacts of MTR/VF on streams. However, we expected varying outcomes to litter breakdown rates depending upon overriding or additive effects on leaf breakdown caused by interactions between abiotic and biotic factors and MTR/VF stressors (Fig. 1). For example, breakdown rates might be slower in VF catchments than in forested catchments because elevated total dissolved ions and metal precipitates impair microbial and macroinvertebrate communities or interfere with leaf colonization and consumption. Alternatively, breakdown rates might be faster in VF constructed channels than in forested catchments because flashier flows could cause high physical fragmentation (i.e., higher abrasion and lower retention).

Methods

Study area

The study sites were in 6 headwater catchments of Buckhorn Creek in Breathitt County in east-central Kentucky, USA. The study area is in the Eastern Coalfield physiographic region and the Central Appalachian ecoregion (level III; Woods et al. 2002). This area is characterized by sandstone, siltstone, shale, and coal geology with mixed mesophytic forest. The dominant tree species on lower catchment slopes were *Quercus alba*, *Liriodendron tulipifera*, *Tsuga canadensis*, and *Fagus grandifolia*, whereas *Quercus velutina* and *Quercus prinus* dominated the upper slopes (Phillippi and Boebinger 1986). The dominant vegetation growing on the VFs were *Lespedeza cuneata* and grasses, with scattered young stands of *Platanus occidentalis*, *Elaeagnus angustifolia*, *Robinia pseudoacacia*, and *Pinus strobus* along the constructed channels. Annual precipitation in the region ranges from 100 to 125 cm, and mean annual air temperature is 12°C (Owenby et al. 2001). Soils are predominantly fine-loamy to loamy-skeletal in texture and are derived from weathered sandstone, siltstone, and shale (Hayes 1998).

Study design

Catchments were classified into 2 treatments: forested and VF. Two catchments, F1 and F2, were completely forested, and 4 catchments, VF1, VF2, VF3, and VF4, contained VFs (4–10 y old) with perimeter

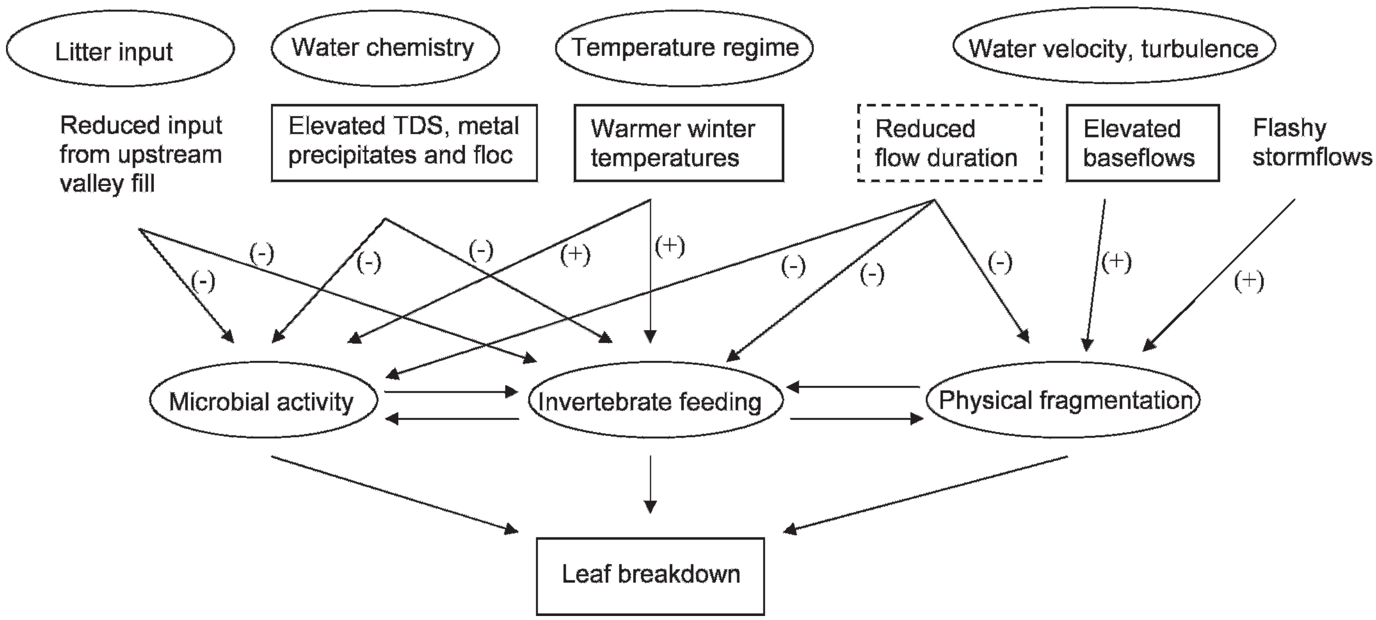

FIG. 1. Factors controlling leaf breakdown (modified from Royer and Minshall 2003) and the expected impact (positive or negative) of mountaintop removal and valley fill (MTR/VF) stressors on leaf breakdown in streams downstream of VFs and in MTR/VF constructed channels on VFs. Stressors in solid boxes are restricted to reaches downstream of VFs and those in dashed boxes are restricted to constructed channels on VFs.

drains. Study reaches (30 m) within each catchment were longitudinally positioned to correspond with documented or expected hydrological permanence classes (Table 1). Ephemeral reaches flow for short durations and only after rain or snow melt because the streambed surface is always above the groundwater table. Intermittent reaches flow seasonally when the groundwater table elevation is above the streambed surface. Perennial reaches have flow throughout most years. Two VF catchments (VF1 and VF2) had intact, natural ephemeral channels upslope of the VFs, whereas the remaining intermittent and ephemeral channels in the VF catchments had constructed channels (Table 1). All perennial reaches in VF catchments were downstream of the VFs and, with the exception of 1 reach (VF2c-P), were not constructed channels. VF2c-P was immediately downstream from VF2 and 130 m upstream of VF2-P, but was within a ~150-m stretch of constructed channel. Riparian trees were sparse to absent along the bank of the constructed intermittent channels adjacent to the VF and were absent along both sides of constructed ephemeral VF reaches. Riparian vegetation was intact along both banks of the perennial reaches downstream of VFs, except VF2c-P where near-bank trees were absent along 1 bank.

Habitat and litter standing crop

The channel habitat was characterized by measuring selected hydrologic, physicochemical, and geomorphologic features of each study reach. The frequency and duration of dry periods were measured with electrical resistance loggers (Fritz et al. 2006). The loggers record binary state (dry or wet) changes at contact ends of a cable, where the presence or absence of water results in a closed or open circuit, respectively. Temperature was measured at 4-h intervals with StowAway TidbiT® temperature loggers (Onset® Computer Corp., Bourne, Massachusetts). Specific conductance and pH were measured with a water-quality sonde (Quanta HydroLab®, Hach Company, Loveland, Colorado) when reaches had flowing surface water. Water samples were collected from perennial stream locations on a biweekly basis (January–July 2006). Sampling, preservation, and analytic protocols were done in accordance to standard procedures (APHA 1992). Samples were filtered through a 0.45-μm syringe filter prior to analysis. SO_4^{2-} and Cl^- concentrations were determined by means of a quantitative ion chromatography procedure on a Dionex Ion Chromatograph (IC) 2000 (Dionex Corporation, Sunnyvale, Califor-

TABLE 1. Characteristics of the 19 study reaches in east-central Kentucky. F = forested, VF = valley fill.

Study reach	Reach permanence	Catchment class (fill volume, m^3)	Channel	Catchment area (ha)	Permit number	Year fill completed
F1-E	Ephemeral	Forested	Natural	2.1	na	na
F1-I	Intermittent	Forested	Natural	19.7	na	na
F1-P	Perennial	Forested	Natural	88.0	na	na
F2-E	Ephemeral	Forested	Natural	1.8	na	na
F2-I	Intermittent	Forested	Natural	10.1	na	na
F2-P	Perennial	Forested	Natural	75.7	na	na
F3-E	Ephemeral	Forested	Natural	1.7	na	na
VF1-I	Intermittent	VF	Constructed	19.0	813-0180	2000
VF1-P	Perennial	VF (1,862,801)	Below fill, natural	42.9	813-0180	2000
F4-E	Ephemeral	Forested	Natural	1.9	na	na
VF2-I	Intermittent	VF	Constructed	15.6	813-0205	1998
VF2c-P	Perennial	VF	Below fill, constructed	33.5	813-0205	1998
VF2-P	Perennial	VF (1,660,218)	Below fill, natural	37.3	813-0205	1998
VF3-E	Ephemeral	VF	Constructed	10.2	813-0156	1995
VF3-I	Intermittent	VF	Constructed	12.2	813-0156	1995
VF3-P	Perennial	VF (9,330,515)	Below fill, natural	45.8	813-0156	1995
VF4-E	Ephemeral	VF	Constructed	0.1	813-0180	2001
VF4-I	Intermittent	VF	Constructed	20.4	813-0180	2001
VF4-P	Perennial	VF (4,509,936)	Below fill, natural	44.1	813-0180	2001

nia). Measurements of Ca^{2+} and Mg^{2+} concentrations were made with a GBC SDS 270 Atomic Adsorption Spectrophotometer (AAS) (GBC Scientific Equipment, Hampshire, Illinois). An ICP-OES - Varian Vista-Pro CCD Simultaneous (Varian Instruments, Palo Alto, California) was used to measure dissolved Fe and Mn. Median particle size, channel slope, canopy cover, and frequency of erosional–depositional sequences were measured once for each reach with methods described by Fritz et al. (2006). Water depth and active channel width were measured on each visit (Fritz et al. 2006). The RBP habitat assessment form for high gradient streams (Barbour et al. 1999) was used to score each site once.

In-channel standing crop of coarse benthic organic matter (CBOM) was measured in October, January, April, and August. All surface and partially buried CBOM was collected by hand along 3 random transects (area = 20 cm long × channel width) spanning the active channel. On occasions when accumulations of CBOM were large, transects were subsampled randomly. CBOM samples were taken to the laboratory, dried (70°C) for >48 h, weighed, combusted at 550°C for ≥2 h, and reweighed to determine ash-free dry mass (AFDM). Each transect was treated as a subsample, and the mean of the 3 transects was the statistical unit for comparisons.

Litter breakdown and invertebrate colonization

Breakdown rates of *Q. alba* (white oak) were determined with a standard litterbag technique (Boulton and Boon 1991). Abscised white oak leaves were collected in aerial littertraps in September–October 2005, pooled among traps and air dried (~20°C) in the laboratory for ~30 d. We placed ~5.0 g of leaves (4.2 g AFDM) in 30 × 35 cm nylon bags with 6-mm mesh size (Hubert Company, Harrison, Ohio). Bags were staked to the streambed surface in pools throughout the study reaches at the end of October 2005 (228 total). Pools were chosen because they were common habitat units and retained surface water longer than other habitat units in our study reaches. Three litter bags were collected from each study reach at time 0 (to estimate handling loss), 21, 82, 166, and 306 d. Upon collection, litter bags were placed individually into resealable plastic bags, stored on ice, and returned to the laboratory.

In the laboratory, litter bag contents were rinsed gently with tap water into a 250-μm sieve to separate oak leaves from fine particulate organic matter and invertebrates. Invertebrates were placed into Whirl-Pak® bags and preserved with 75% ethanol prior to identification, measurement, and enumeration. Both aquatic and terrestrial invertebrates were included in our study because ephemeral and intermittent reaches regularly dry and are colonized by aquatic and terrestrial fauna that use organic matter as a resource. Most aquatic taxa were identified to genus (except chironomids [to tribe], mites and oligochaetes [to family], and meiofauna [to suborder, order, or phylum]), whereas terrestrial insects, snails, and spiders were identified to family and other terrestrial taxa to order or suborder (e.g., Diplopoda, Pseudoscorpiones, Oribatida). Because of the limited taxonomic resolution, all nematodes were assumed to be aquatic and mites that were not hydracarina (e.g., Oribatida, Gamasida) were assumed to be terrestrial. Shredder biomass was estimated using published allometric equations (e.g., Edwards 1967, Sample et al. 1993, Benke et al. 1999).

Subsamples (20) were taken with a cork borer (9.5-mm diameter) from randomly selected leaves in each litter bag, blotted, weighed, placed in methanol, and stored in a freezer until analysis for ergosterol concentration (Montgomery et al. 2000). Ergosterol concentration in each sample was measured using an HP series 1100 HPLC high performance liquid chromatograph (Hewlett Packard, Palo Alto, California) with a Varian Microsorb MV (100 Angstroms) column (Varian Instruments, Palo Alto, California). Ergosterol was expressed as μg/g ash-free dry mass (AFDM) litter. The remaining leaf material was dried at 70°C for >48 h, weighed, and ground to a fine powder in a mill. Separate subsamples of the ground material were weighed and used to determine % AFDM and C to N ratio (C:N). Subsamples were combusted at 550°C for 2 h to determine % AFDM. Total C (organic and inorganic) and N contents of subsamples were determined by dry combustion in a LECO CHN 2000 analyzer (LECO Corporation, St. Joseph, Michigan).

Data analysis

Concentrations of dissolved SO_4^{2-}, Cl^-, Mn, Mg^{2+}, Fe, and Ca^{2+} at perennial sites were compared between VF and forested catchments with Mann–Whitney *U* tests. Habitat measures were compared across catchment (forested and VF) and permanence (ephemeral, intermittent, and perennial) classes with a standardized or correlation-based principal components analysis (PCA; PC-ORD, version 4.25; MjM Software, Gleneden Beach, Oregon). The habitat variables included mean duration of dry periods, the frequency of dry periods, channel slope, median sediment particle size, mean water depth, mean active channel width, frequency of erosional–depositional sequences, mean canopy cover, cumulative degree days, and RBP scores. Variables were transformed prior to analysis if they were not normally distributed.

Breakdown rates were calculated only through 166 d, rather than the full 306 d because of litterbag losses at several sites. Percent AFDM remaining was used to calculate a breakdown rate (k) for each litter bag based on the formula, breakdown rate = (ln[final AFDM/initial AFDM])/cumulative degree days) (Huryn et al. 2002). We also calculated breakdown rate on a per day basis because MTR/VF might alter water temperature. Litter breakdown rates (per day and per degree day) were compared across perennial reaches to determine if rates at the constructed perennial reach differed from the natural perennial reaches (PROC GLM, SAS 9.1; SAS Institute, Cary, North Carolina). For this initial comparison, each final litter bag within a reach was treated as a replicate. Breakdown rates were compared across catchment and permanence classes with a 2-way analysis of variance (ANOVA; PROC GLM). For this analysis the average breakdown rate across litter bags from a reach was treated as a replicate.

Standing crop of CBOM was compared across catchment and permanence classes with a repeated measures 2-way ANOVA (PROC MIXED with Kenward–Rogers adjustment for degrees of freedom). Variables associated with litter bags included ergosterol concentration, C:N of litter, total invertebrate density (number of invertebrates/g litter remaining), total invertebrate taxon richness, aquatic invertebrate density, aquatic taxon richness, taxon richness within insect orders Ephemeroptera, Plecoptera, and Trichoptera (EPT richness), and shredder density. Mean values across litter bags collected from a reach at a time period for these variables were treated as the statistical unit for comparisons, and values were compared across catchment and permanence classes with a repeated measures 2-way ANOVA (PROC MIXED with Kenward–Rogers adjustment for degrees of freedom). For all repeated measures analyses, the best fit covariance structure was selected based on relevance to our design and corrected Akaike Information Criteria (Wang and Goonewardene 2004). Multiple comparison tests (LSMEANS, Tukey adjustment) were used to determine where specific differences resided if significant differences were found among treatments.

Invertebrate assemblage structure was compared based on nonmetric multidimensional scaling (NMS, PC-ORD version 5.1) and analysis of similarities (ANOSIM; PRIMER version 5.2, PRIMER-E, Plymouth, UK). Invertebrate assemblage structure across samples was visually assessed with NMS, an ordination technique (Clarke and Warwick 2001). This technique reduces the complexity associated with data (multiple species across many sites) to a few axes that might capture variation across study reaches. The number of taxa was reduced from 114 to 74 by using only those taxa that had >4% relative abundance in at least 1 litter bag (removing 214 of 9689 individuals). Abundance was averaged across litter bags collected from each site and date resulting in a matrix of 74 taxa and 71 samples (5 sites had no litter bags remaining at

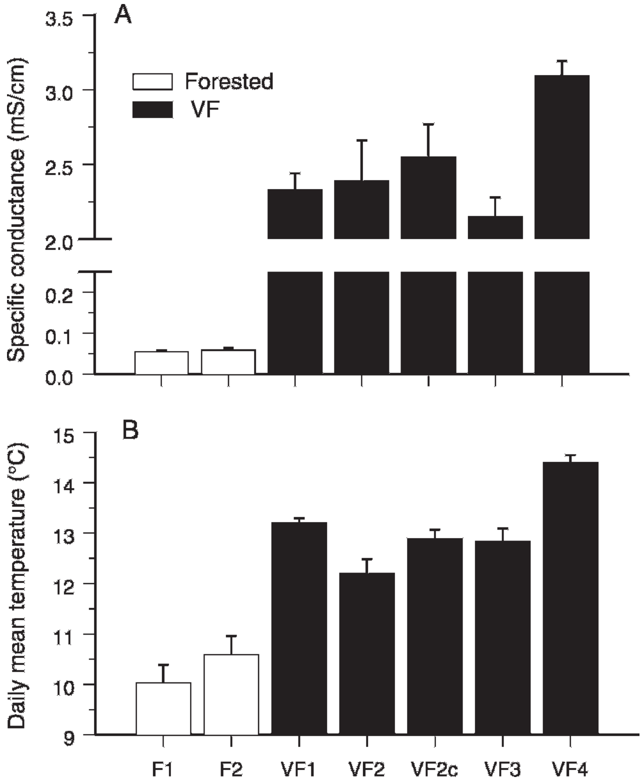

FIG. 2. Mean (+1 SE) specific conductance (A) and daily mean water temperature (B) at perennial sites over the study period. VF = valley fill, F = forested.

the 306 d collection). The data were 4th-root transformed (Field et al. 1982), and the Bray–Curtis coefficient was used as the distance measure in the NMS ordination. The dimensionality of the final ordination was determined by Monte Carlo simulations (99 runs) and Shepard plots. We used 2-way crossed ANOSIM to test the hypothesis that assemblage structure does not differ significantly among groups of litter bags (catchment and permanence classes). This test compares the test statistic, R (calculated difference between the rank similarities of assemblages between and within groups) to a null distribution of random R values derived from permutations (999) of the data. The typical range of R values is from 0 (assemblages not different) to 1 (assemblages completely different). The individual contributions of taxa to dissimilarity among litter bag assemblages were identified using similarity percentage-species contributions (SIMPER; PRIMER). Relationships between breakdown rate and structural measures (PCA scores, mean ergosterol concentration, CBOM standing crop, mean shredder biomass, and RBP scores) were assessed using Spearman rank correlations.

TABLE 2. Mean (±1 SE) dissolved concentrations of dominant ions in perennial streams at valley fill and forested sites. Samples were collected biweekly from January to July 2006. Mann–Whitney U tests were done to compare valley fill and forested sites.

	Concentration (mg/L)		
Ion	Valley fill	Forested	p
Cl^-	2.6 (0.4)	0.8 (0.2)	<0.001
SO_4^{2-}	1187 (85)	11 (1.7)	<0.001
Mg^{2+}	196.0 (15.0)	2.7 (0.2)	<0.001
Ca^{2+}	92.0 (13.0)	9.8 (3.7)	<0.001
Fe	6.9 (1.8)	0.1 (0.03)	<0.001
Mn	29.0 (4.6)	0.2 (0.06)	<0.001

Results

Specific conductance was, on average, 36 to 57× higher in perennial reaches below VFs than at perennial forested sites (Fig. 2A), whereas pH ranged from 6.0 to 7.2 and was similar among perennial reaches downstream of VFs and in perennial forested reaches. Daily mean water temperatures were 2 to 4°C higher in perennial streams downstream of VFs than in perennial forested reaches (Fig. 2B) and dissolved O_2 (range: 6.1–12.9 mg/l) varied as much over the study visits as among streams. Dissolved concentrations of SO_4^{2-}, Cl^-, Mn, Mg^{2+}, Fe, and Ca^{2+} were significantly higher in perennial reaches downstream of VFs than in perennial forested reaches (Table 2). The proportion of time intermittent and ephemeral channels were dry was, on average, higher for the constructed channels (intermittent = 0.62; ephemeral = 0.999) draining VFs than for natural channels (intermittent = 0.11; ephemeral = 0.88) draining forest. Habitat characteristics varied across catchment and permanence classes (Fig. 3). The first 2 PCA axes explained 67% of the variation in habitat variables among sites, whereas the remaining PCA axes each explained <10% of the variation. The variables with highest loadings on PC1 were water depth (r = 0.86) and channel slope (r = −0.85), whereas degree days (r = −0.88) and RBP scores (r = 0.73) had the highest loadings on PC2. Among permanence classes, the largest difference in ordination space was between intermittent sites in forested and VF catchments. The habitat of intermittent forested sites was more comparable to habitat of perennial sites than habitat of intermittent VF sites, which plotted nearer to ephemeral sites (Fig. 3).

CBOM standing crop did not differ across time periods (Table 3). CBOM varied across catchment and permanence classes, and the interaction term was significant (Fig. 4). Ephemeral forested reaches had significantly higher CBOM than constructed ephem-

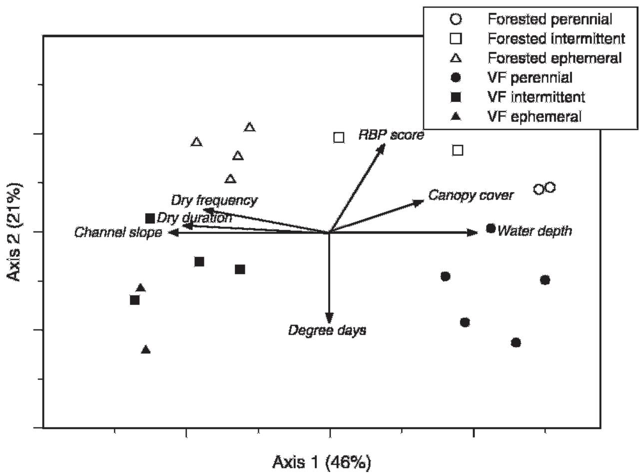

FIG. 3. Ordination from principal component analysis (PCA) of habitat variables from the 19 study sites. Arrows represent linear correlations ($r > \pm 0.7$) between PCA scores and the individual habitat variables. Numbers in parentheses after axis titles are % total variance explained by the axis. VF = valley fill, RBP = Rapid Bioassessment Protocol.

eral VF reaches (adjusted Tukey's test, $p < 0.0001$), whereas CBOM in intermittent ($p = 0.07$) and perennial ($p = 0.5$) sites did not differ significantly between catchment treatments. CBOM also varied longitudinally in forested and VF catchments. Ephemeral reaches (located furthest upstream) in forested catchments tended to have higher levels of CBOM than reaches further downstream. The reverse pattern was seen in VF catchments, where perennial reaches (furthest downstream) typically had higher CBOM levels than reaches further upstream.

Breakdown rates (per day or per degree day) at the perennial reach with a constructed channel (VF2c-P) did not differ from rates in other perennial VF reaches ($p > 0.41$, adjusted Tukey's test). Therefore, we treated VF2c-P as a replicate perennial VF reach in the subsequent analyses. The overall model comparing breakdown rates across catchment and permanence classes was significant whether calculated per day ($F_{5,18} = 25.83$, $p < 0.0001$) or per degree day ($F_{5,18} = 20.99$, $p < 0.0001$). Breakdown per day differed for catchment class ($F_{1,13} = 61.26$, $p < 0.0001$), permanence class ($F_{2,13} = 33.45$, $p < 0.0001$), and their interaction ($F_{2,13} = 12.42$, $p = 0.001$). Breakdown per degree day also differed for catchment class ($F_{1,13} = 45.43$, $p < 0.0001$), permanence class ($F_{2,13} = 17.60$, $p = 0.0002$), and their interaction ($F_{2,13} = 14.13$, $p = 0.0005$). Breakdown rates at intermittent and perennial reaches in forested catchments did not differ per day (adjusted Tukey's test, $p = 0.08$) or per degree day ($p = 0.71$), but were significantly faster than rates in

TABLE 3. *F*-values (numerator, denominator degrees of freedom) for repeated measures analysis of variance (ANOVA) (PROC MIXED) comparing litterbag variables across treatments. Treatment effects (fixed) were catchment (C; forested and valley fill) and permanence (P; perennial, intermittent, and ephemeral). Time (T) was treated as random variable in the model. The best-fit covariance structure is shown in parenthesis below variable names (VC = variance components, CSH = heterogeneous compound symmetry, ANT1 = 1[st]-order antedependence, and UN = unstructured). EPT = Ephemeroptera, Plecoptera, Trichoptera, CBOM = coarse benthic organic matter, * = $p \leq 0.05$, ** = $p \leq 0.01$, *** = $p \leq 0.001$.

Effect	CBOM standing crop[a] (VC)	C:N[a] (ANT1)	Ergosterol[b] (CSH)	Total density[a] (ANT1)	Aquatic density[a] (CSH)	Shredder density[b] (ANT1)	Total richness[c] (VC)	Aquatic richness[a] (VC)	EPT richness[c] (CSH)
C	20.89*** (1,64)	0.00 (1,13.6)	9.76** (1,24.3)	12.32** (1,19.8)	20.39*** (1,37.5)	23.88*** (1,19.2)	22.11*** (1,59)	26.85*** (1, 9)	85.37*** (1,32.2)
P	5.25** (2,64)	1.42 (2,13.6)	6.97** (2,24.3)	7.54** (2,19.8)	26.87*** (2,37.5)	6.59** (2,19.2)	7.91*** (2,59)	21.35*** (2,59)	17.58*** (2,32.2)
T	2.17 (1,64)	37.38*** (1,12.6)	14.49** (1,9.6)	11.06** (1,20.5)	32.30*** (1,29.7)	9.96** (1,16.3)	24.98*** (1,59)	15.40*** (1,59)	1.39 (1,11.9)
C × P	7.36** (2,64)	0.08 (2,13.6)	5.79** (2,24.3)	3.01 (2,19.8)	6.90** (2,37.5)	6.82** (2,19.2)	5.45** (2,59)	8.45*** (2,59)	17.02*** (2,32.2)
C × T	0.05 (1,64)	0.05 (1,12.6)	28.90*** (1,9.6)	3.33 (1,20.5)	5.96* (1,29.7)	1.98 (1,16.3)	0.58 (1,59)	1.60 (1,59)	2.33 (1,11.9)
C × P × T	0.24 (2,64)	0.02 (2,12.6)	12.09** (2,9.6)	0.79 (2,20.3)	0.80 (2,29.5)	0.02 (2,16.2)	3.00 (2,59)	0.53 (2,59)	0.43 (2,12.0)
C × P × T	0.24 (2,64)	0.02 (2,12.6)	12.09** (2,9.6)	0.79 (2,20.3)	0.80 (2,29.5)	0.02 (2,16.2)	3.00 (2,59)	0.53 (2,59)	0.43 (2,12.0)

[a] $\log(x+1)$ transformed
[b] $x^{0.25}$ transformed
[c] $\sqrt{(x)}$ transformed

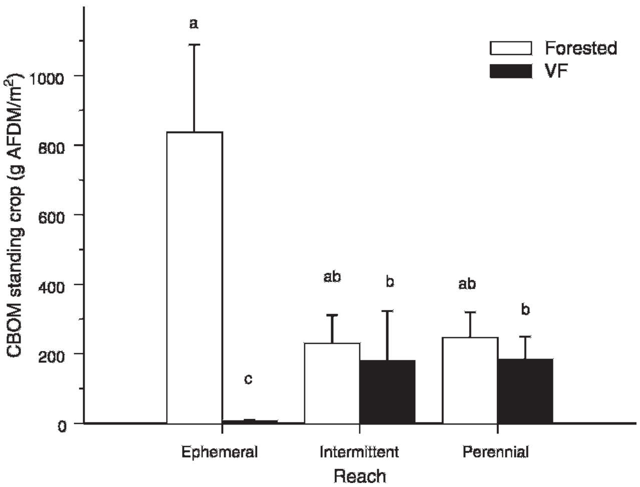

FIG. 4. Mean (+1 SE) standing crop of coarse benthic organic matter (CBOM) in active channels at ephemeral, intermittent, and perennial study sites for the 4 sampling dates. Bars with the same letters are not significantly different ($p > 0.05$). AFDM = ash-free dry mass, VF = valley fill.

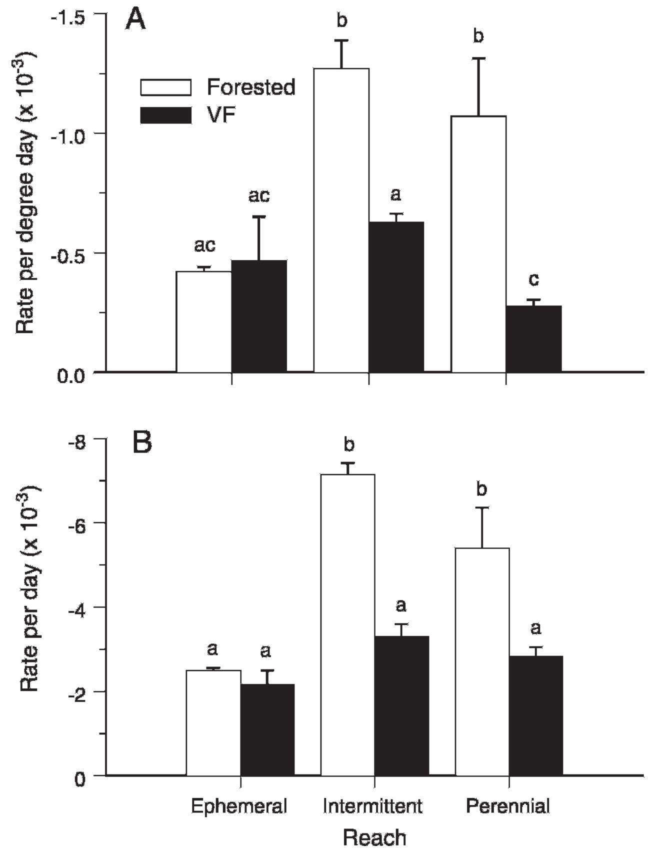

FIG. 5. Mean (+1 SE) breakdown rates of oak leaves per degree day (A) and per day (B) over 166 days (October–April) at ephemeral, intermittent, and perennial study sites. Bars with the same letter were not significantly different ($p > 0.05$). VF = valley fill.

all VF reaches and at ephemeral reaches in forested catchments (Fig. 5A, B). Breakdown rates per degree day at intermittent VF reaches (constructed channels) were faster than those at perennial VF reaches (natural channels below VFs; $p = 0.02$).

Litter C:N did not differ between catchment classes or across permanence classes, but C:N did decline over time across all sites (slope = -0.09, $R^2 = 0.41$, $p < 0.0001$, $n = 197$; Table 3). Ergosterol concentration of leaves did not differ consistently between catchment classes or among permanence classes, and several interaction terms were significant (Table 3, Fig. 6A–C). Ergosterol concentrations increased over time at ephemeral (Fig. 6A) and perennial reaches (Fig. 6C), but this trend was not apparent across all intermittent reaches (Fig. 6B).

All measures of invertebrate density and richness varied between catchment classes (Table 3, Figs 7A–F, 8A–F). Total density in litter bags from forested catchments was significantly higher than in litter bags from VF catchments (adjusted Tukey's test, $p < 0.05$), regardless of permanence (Fig. 7A–C). The remaining measures were significantly higher at perennial and intermittent sites from forested catchments than at perennial and intermittent sites from VF catchments, but did not differ between ephemeral sites in forested and VF catchments (e.g., shredder density; Fig. 7D–F). All measures were higher at intermittent reaches in forested catchments than at perennial reaches in VF catchments. Litter bags from forested ephemeral reaches had higher total richness and aquatic richness (not shown) than litter bags from VF perennial reaches (Fig. 8A–C). No invertebrate measures differed at intermittent and perennial reaches in forested catchments. In contrast, aquatic density (not shown), total richness, and aquatic richness were higher in litter bags at perennial than at intermittent reaches in VF catchments.

All invertebrate measures, except EPT richness, varied with time and most measures (particularly density) tended to increase over the study period (Table 3, Figs 7A–F, 8A–F). A significant interaction between catchment and time indicated that litter bag densities of aquatic invertebrates varied over time in forested catchments, but not in VF catchments.

Invertebrate assemblages were weakly clustered by catchment and permanence classes in a 2-dimensional plot of the NMS ordination (stress = 19.4; Fig. 9). These patterns were supported by ANOSIM results, where the invertebrate assemblage structure within litter bags varied significantly but not strongly by catchment and permanence class (based on moder-

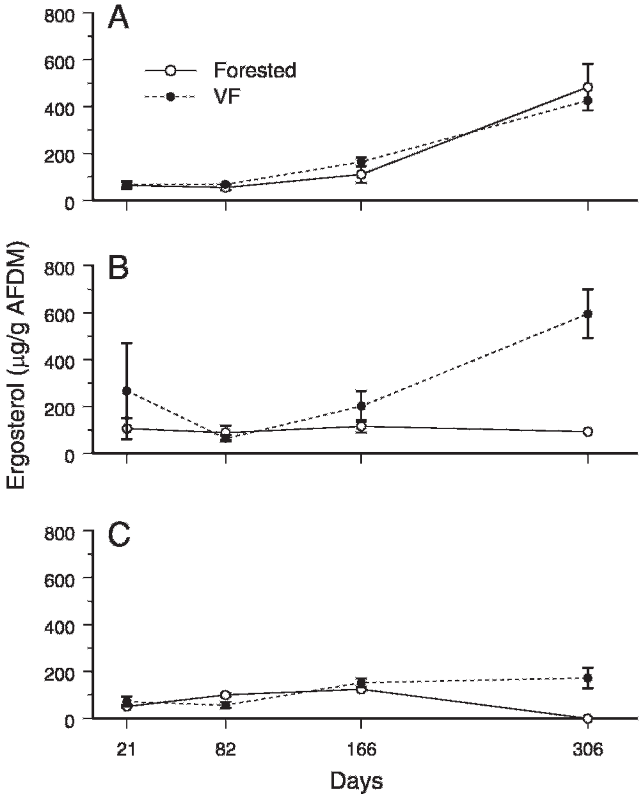

FIG. 6. Mean (±1 SE) ergosterol concentration for oak leaves at ephemeral (A), intermittent (B), and perennial (C) study sites in forested and valley fill (VF) catchments. Leaf material remained at 1 of the perennial forested sites in August (306 d) and had undetectable ergosterol. AFDM = ash-free dry mass.

ately low R values; Table 4). The NMS 1st axis scores were most strongly correlated with PC1 (representing hydrology and channel slope; $r = -0.52$), whereas catchment area ($r = -0.40$) and PC2 (representing degree days and RBP scores; $r = -0.36$) were the environmental variables most strongly correlated with NMS 2nd axis scores. The assemblages with the highest overlap were from intermittent and perennial reaches in forested catchments. Intermittent and ephemeral reaches were not clearly separated in the VF catchments.

Ninety percent of the overall dissimilarity between assemblages from forested and VF catchments was accounted for by 46 different taxa, and no individual taxon contributed >10%. Chironomid (Tanytarsini, Orthocladiini, Corynoneurini) tribes, oligochaete (Naididae and Enchytraeidae) families, isotomid collembolans, and *Eurylophella funeralis* (Ephemeroptera: Ephemerellidae) were the top contributing taxa to the dissimilarity between forested and VF assemblages. Of the 46 taxa, only Ostracoda, *Culicoides* spp., Diplopoda, Aphididae, Formicidae, and Thripidae were more abundant in VF than in forested catchments. No Ephemeroptera were collected from any VF site, whereas Ephemeroptera were present in 94, 85, and 0% of the litter bags retrieved from perennial, intermittent, and ephemeral forested reaches, respectively.

Breakdown rates per degree day were not correlated with RBP scores (Fig. 10A), PC1 scores, or CBOM standing crop. The correlation between PC2 and litter breakdown rates was not assessed because degree day was strongly correlated with PC2 and was incorporated in our calculations of breakdown rates. Breakdown rates were correlated with shredder richness ($r_S = 0.58$, $p = 0.009$, $n = 19$; Fig. 10B) but not biomass ($r_S = 0.38$, $p = 0.10$, $n = 19$; Fig. 10C) in the leaf packs, although shredder metrics were correlated with one another ($r_S = 0.86$, $p < 0.0001$, $n = 19$). Specific conductance was strongly negatively correlated with breakdown rates at perennial and intermittent sites ($r_S = 0.90$, $p = 0.0003$, $n = 10$; Fig. 10D), as were Fe, Cl^-, SO_4^{2-}, and Mn.

Discussion

Oak leaves broke down faster at perennial and intermittent reaches in forested catchments than at reaches in VF catchments, regardless of whether rates were measured per day or per degree day. This result suggests that alteration of thermal regime was not the primary driver of the difference in leaf breakdown between VF and forested catchments. We did not detect consistent differences in ergosterol concentration between catchment treatments. Therefore, the differences in breakdown rate might have been driven by the large differences in invertebrate assemblages colonizing litter bags or differences in microbial activity not reflected in ergosterol concentrations. The influence of invertebrate assemblages was further supported by the correlations between leaf breakdown rates and shredder richness. However, because of the extreme alteration of ecosystems by MTR/VF, it is likely that multiple mechanisms with varying degrees of influence contributed to the differences in litter breakdown between catchment classes. *Acer rubrum* leaves broke down more slowly and had fewer invertebrates in a stream draining a VF in Maryland, USA, than leaves in a nearby reference stream (Simmons et al. 2008). However, unlike in our study, the Maryland VF stream was intermittent, and the reference stream was perennial. Based on the flashy hydrology and tattered appearance of the litter upon collection (particularly at intermittent VF reaches) in our study, we suspect that breakdown in the constructed channels was driven primarily by physical

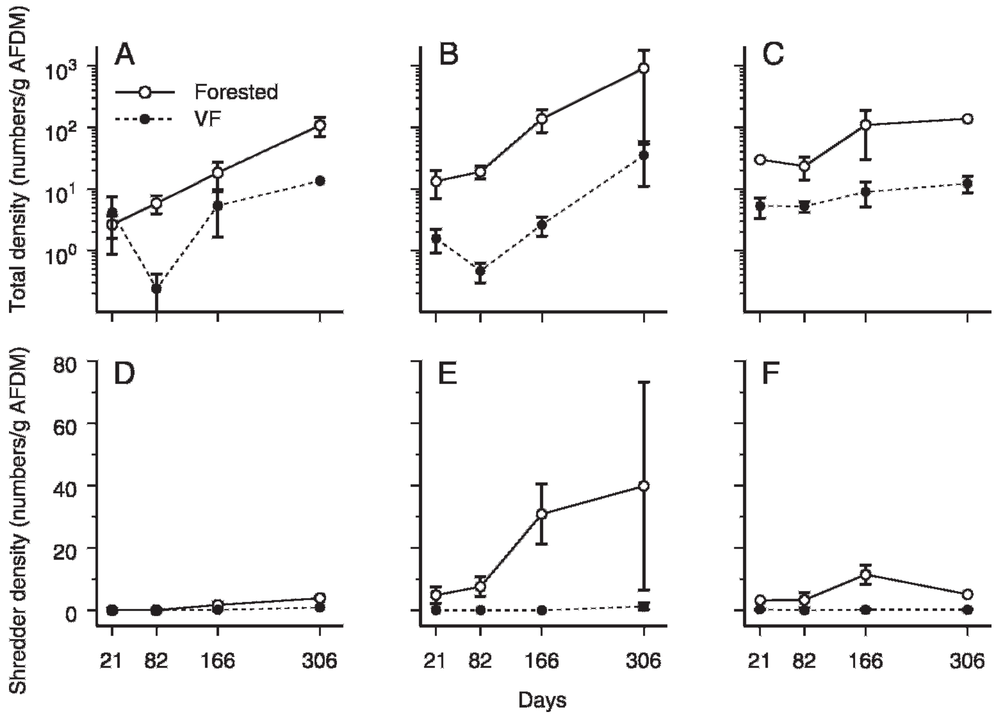

FIG. 7. Mean (±1 SE) total invertebrate density in leaf litter bags at ephemeral (A), intermittent (B), and perennial (C) study reaches and shredder density in litter bags in ephemeral (D), intermittent (E), and perennial (F) study reaches in forested and valley fill (VF) catchments. AFDM = ash-free dry mass.

processes (i.e., abrasion, flow-related fragmentation). The relatively high invertebrate densities in litter bags at intermittent and perennial forested sites compared to those at VF sites also suggest that stream invertebrates were largely responsible for differences in breakdown rates between catchment classes.

Gessner and Chauvet (2002) developed a framework for classifying stream functional integrity based on litter breakdown rates. They recognized that different stressors can have different effects on breakdown rates, i.e., some might increase or decrease breakdown relative to breakdown in reference sites. The ratio of breakdown rate (per degree day) at each VF site and the average breakdown rate across forested sites and the ranges provided by Gessner and Chauvet (2002) can be used to classify ecosystem function of the VF reaches as: 1) no clear evidence of impact, 2) mildly affected, or 3) severely compromised. Based on the breakdown ratios for oak leaves, all perennial and 2 intermittent VF sites (VF1-I and VF3-I) were severely compromised (ratios < 0.5), whereas the ephemeral and the other 2 intermittent VF sites were mildly affected (0.5–0.75 or 1.33–2.0).

Iron precipitates and flocs are common in streams draining coal mines because of the frequent geologic association of iron ores and coal (Kelly 1988). These precipitates also might have contributed to differences among perennial sites. Surface-water pH was circumneutral, but iron hydroxide precipitates (i.e., ochre) blanketed the stream bed and cemented together underlying stones at all perennial VF sites. Ochre was not present at perennial forested sites. At the perennial VF sites, flocs of Fe-depositing bacteria (Ghiorse 1984) were present at the start of the study, were apparently washed out during winter, and returned during the summer. The combination of high Fe content, the neutralizing effect of the overburden, and riparian shading probably lead to precipitation of Fe in the perennial reaches below VFs. Iron precipitates and flocs can have direct and indirect effects on stream invertebrates through toxicity, by inhibiting movement, respiration, and feeding, and by altering the benthic environment (Vuori 1995). The Fe probably contributed to lower breakdown rates and abundance and diversity of invertebrates in litter bags at the VF perennial sites compared to in litter bags at forested perennial sites. Authors of several studies of the impact of mine drainage on streams have reported slower breakdown rates and microbial activity with increasing levels of metal concentrations or precipitates (Gray and Ward 1983, Bermingham et al. 1996, Niyogi et al. 2001, 2002, Siefert and Mutz 2001, Schlief 2004, but see Barnden and Harding 2005). Remediation of mine drainage

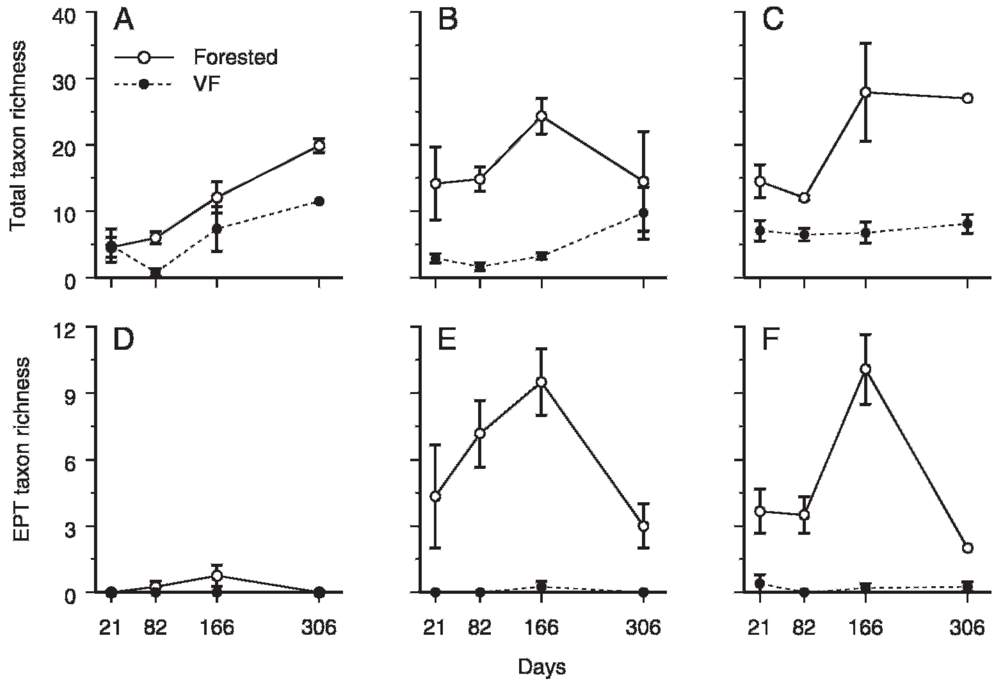

FIG. 8. Mean (±1 SE) total taxon richness in leaf litter bags at ephemeral (A), intermittent (B), and perennial (C) study reaches and Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxon richness in litter bags in ephemeral (D), intermittent (E), and perennial (F) study reaches in forested and valley fill (VF) catchments.

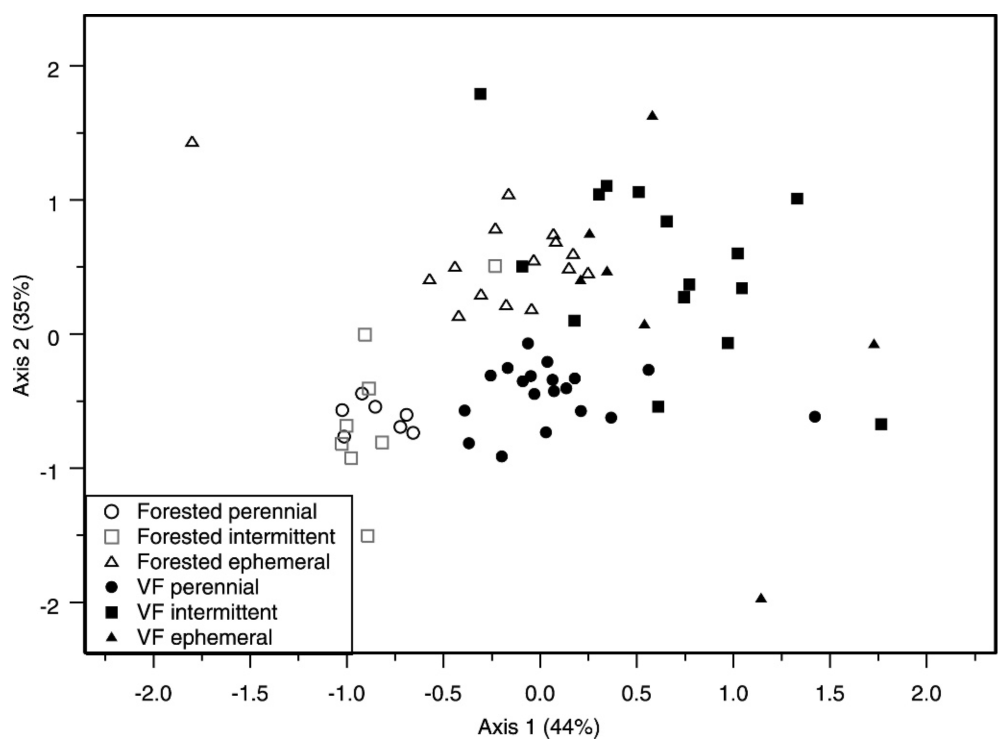

FIG. 9. Ordination from nonparametric multidimensional scaling (NMS) of invertebrate assemblages in litter bags (4th-root transformed abundance). Each symbol represents the average of litter bags collected at a study site on a date. Numbers in parentheses after axis titles are % total variance explained by the axis. Stress for 2-dimensional solution = 19.4. VF = valley fill.

TABLE 4. Results of 2-way analysis of similarities (ANOSIM) tests for invertebrate assemblages in litter bags across catchment and permanence classes. $n = 71$. VF = valley fill.

Test	Factor	Pairwise comparison	R	p
Catchment × permanence	Catchment	Global (forested vs VF)	0.56	0.001
	Permanence	Global	0.52	0.001
		Perennial vs intermittent	0.49	0.001
		Perennial vs ephemeral	0.72	0.001
		Intermittent vs ephemeral	0.35	0.001

should redress the effects of both dissolved metals and metal deposition.

Specific conductance and contributing ions (e.g., SO_4^{2-}, Fe) also are typically elevated in streams with mine drainage (Gray and Ward 1983, Bermingham et al. 1996, Siefert and Mutz 2001). However, conductivity and contributing ions associated with other types of mining are rarely as high as is seen in streams draining MTR/VF coal mining. Young et al. (2008) predicted increasing breakdown rates with increasing specific conductance level. Their prediction was based on results of a study that compared pairs of soft- (mean specific conductance: 36 and 38 μS/cm) and hard-water streams (273 and 320 μS/cm), in which higher shredder densities and microbial activity contributed to faster litter breakdown in hard-water streams than in soft-water streams (Rosset et al. 1982). Our result indicates that the relationship between specific conductance and litter breakdown might not always be a simple positive relationship, in which high levels of particular contributing ions or processes (e.g., Fe precipitate, travertine formation; Casas and Gessner 1999) slow litter breakdown. Central Appalachian streams draining MTR/VF with elevated

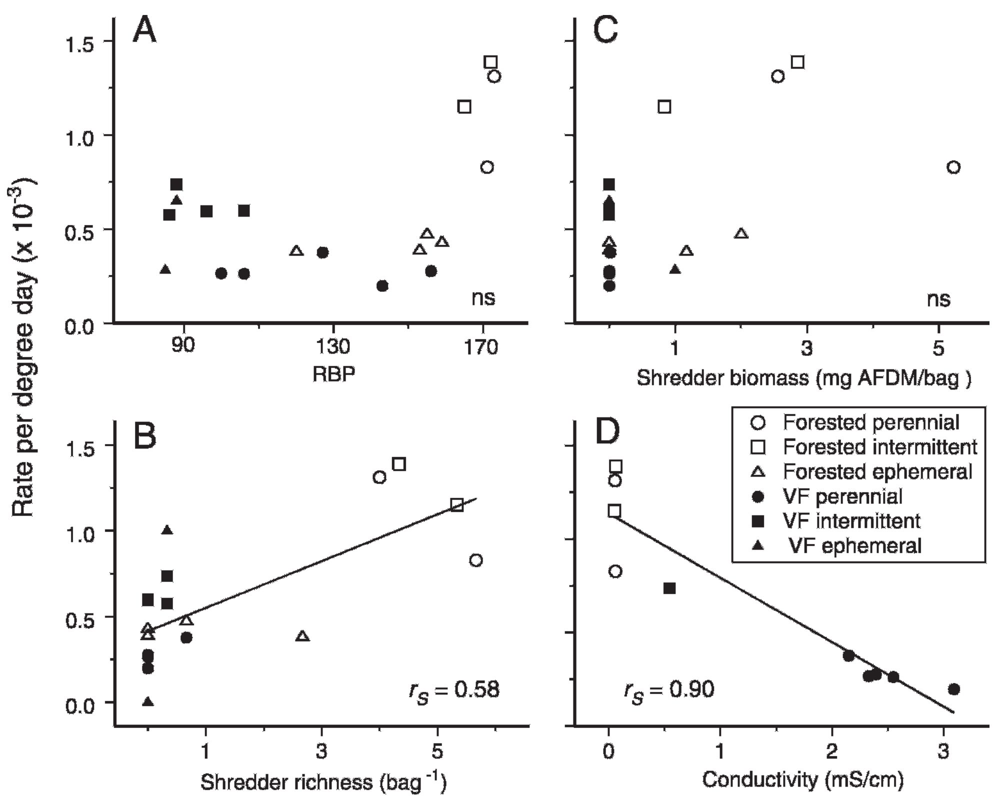

FIG. 10. Mean breakdown rates (per degree day) of oak leaves (over 166 days from October–April) vs Rapid Bioassessment Protocol habitat assessment (RBP) scores ($n = 19$) (A), shredder richness (in litter bags collected at day 166) ($n = 19$) (B), mean shredder biomass (in litter bags collected at day 166) ($n = 19$) (C), and mean annual conductivity at perennial and intermittent reaches based on ≥ 2 measurements of conductivity ($n = 10$) (D). AFDM = ash-free dry mass, VF = valley fill, ns = not significant, r_s = Spearman correlation coefficient.

specific conductance >500 μS/cm have impaired macroinvertebrate assemblages (Hartman et al. 2005, Pond et al. 2008). Therefore, specific conductance might be indirectly related to leaf breakdown rates by affecting invertebrate assemblages. MTR/VF mining is an extreme example, but elevated dissolved solids and specific conductance are associated with most landscape disturbances (e.g., Tuch and Gasith 1989, Dow and Zampella 2000, Paul and Meyer 2001). Further research is needed to characterize better the relationships between total dissolved solids (or specific conductance), chemical constituent composition, aquatic communities, and ecosystem functions.

Recommendations

RBP habitat assessment scores were not correlated with leaf litter breakdown rates, a result that indicates that the RBP does not reflect litter breakdown function in headwater streams. Shredder biomass/richness and specific conductance were strongly correlated with leaf litter breakdown rates. Shredders are more directly linked to leaf breakdown than the stream features included in the RBP, and at least among our perennial sites, specific conductance might be an important factor controlling shredder biomass and diversity. Further understanding of relationships between stream functions and structure is needed to inform appropriate assessment methods fully. The current dependence upon the RBP score to quantify stream function in forested headwater streams is inadequate.

Groin drains are required under SMCRA to prevent destabilization of VFs. However, our findings suggest that these channels should not be considered as on-site mitigation for the natural channels buried under VFs. The habitat features and aquatic assemblages differed greatly between constructed and natural channels. Some USACE districts currently weight mitigation requirements for stream impacts authorized under CWA Section 404 by hydrologic permanence. Litter breakdown and invertebrate assemblages did not vary between forested intermittent and perennial sites in our study. Natural ephemeral channels did contain aquatic life, but they differed to a greater extent in structure and function from downstream perennial reaches than the difference between intermittent and perennial reaches. We recommend that more consideration should be applied to determining the value of intermittent reaches (pre- and post-construction). Specifically, more precise quantification of flow duration (directly or indirectly; e.g., Fritz et al. 2006) should be used to determine mitigation value because flow duration varies greatly among intermittent streams and is an important determinant of structure and function in temporary streams (e.g., Chadwick and Huryn 2007, Larned et al. 2007). Breakdown rates in constructed and natural ephemeral channels did not differ, but substantial evidence (CBOM, invertebrate assemblage) indicated that constructed channels did not adequately replace natural ephemeral channels.

Specific conductance and concentrations of individual ions like SO_4^{2-} and Mn are higher in streams draining MTR/VF than in forested reference streams (Jack et al. 2005, Pond et al. 2008), increase after MTR/VF mining (Jack et al. 2005), and do not decline with VF age (at least for 20 y; Merricks et al. 2007). The expected duration of leaching of dissolved solids that contribute to elevated specific conductance of water originating from VFs is unknown. Impaired osmoregulation, metal toxicity, reduced reproductive success, and altered behavior have been hypothesized as causal mechanisms for altered invertebrate communities (Pond et al. 2008). Experimental studies are needed to identify: 1) the mechanism(s) by which the invertebrate assemblages and their functions are altered, 2) protective criteria for Central Appalachian stream communities, and 3) feasible remediation methods for MTR/VF impacted streams.

Geomorphic alteration and forest fragmentation by MTR/VF mining have altered the landscape of the Central Appalachians (Hooke 1999, Wickham et al. 2007), and the cumulative consequences on downstream waters and the regional climate are unknown. Our study supports previous work that reports alteration of the local aquatic community by MTR/VF. Other studies have reported the impacts of MTR/VF to terrestrial flora and fauna (Holl 2002, Chamblin et al. 2004, Skousen et al. 2006, Wood et al. 2006). Understanding the limitations of altered soil and geologic conditions (compaction, drainage, fertility) and appropriate expectations over time are important steps for the recovery of forests and associated terrestrial communities (Holl 2002, Skousen et al. 2006, Craw et al. 2007). Given the severe alteration to the underlying geology in VFs, it is unclear if aquatic communities adapted to water with low dissolved ion concentrations and the functions they contribute can fully recover from MTR/VF mining, even after recovery of the upland forests. Because of the conservative nature of the dominant ions contributing to elevated conductivity and the absence of appropriate and viable treatment technologies, the only option for protecting mainstem rivers that drain tributaries with MTR/VF mines is dilution from undisturbed forested tributaries. Thus, regulatory agencies and the mining industry must appropriately weigh the long-term cumulative impacts against the short-term economic gain of coal extraction when planning operations and reviewing permit applications.

Acknowledgements

We thank Dean Hardy, Wes Daniel, Richard Pirkle, and Robbie Johnson for help in the field; Margaret Carreiro, Richard Schultz, and Natalie Abram for ergosterol analysis; Sally Maharaj and Millie Hamilton for water-chemistry and CHN analysis; Clyde Cook, Davie Ransdell, and Kentucky Division of Mine Permits staff for help with coal mine permit information; Will Marshall for logistical support; and Karen Blocksom for statistical advice. Michael Griffith, Margaret Palmer, Maggie Passmore, Greg Pond, Brian Topping, Jack Webster, Manuel Graça, and 2 anonymous reviewers provided helpful comments. Although this work was reviewed by USEPA and approved for publication, it might not necessarily reflect official Agency policy. Mention of trade names or commercial products does not constitute endorsement or recommendation for use. This paper is dedicated in memory of Jeff Jack, 1963–2007.

Literature Cited

ANDERSON, S., AND R. NEWELL. 2004. Prospects for carbon capture and storage technologies. Annual Review of Environment and Resources 29:109–142.

APHA (AMERICAN PUBLIC HEALTH ASSOCIATION). 1992. Standard methods for the examination of water and wastewater. 18th edition. American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, DC.

BARBOUR, M. T., J. GERRITSEN, B. D. SNYDER, AND J. B. STRIBLING. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. 2nd edition. EPA/841/B/98–010. Office of Water, US Environmental Protection Agency, Washington, DC.

BARNDEN, A. R., AND J. S. HARDING. 2005. Shredders and leaf breakdown in streams polluted by coal mining in the South Island, New Zealand. New Zealand Natural Sciences 30:35–48.

BENKE, A. C., A. D. HURYN, L. A. SMOCK, AND J. B. WALLACE. 1999. Length–mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. Journal of the North American Benthological Society 18: 308–343.

BERMINGHAM, S., L. MALTBY, AND R. C. COOKE. 1996. Effects of a coal mine effluent on aquatic hyphomycetes. I. Field study. Journal of Applied Ecology 33:1311–1321.

BOULTON, A. J., AND P. I. BOON. 1991. A review of methodology used to measure leaf litter decomposition in lotic environments: time to turn over an old leaf? Australian Journal of Marine and Freshwater Research 42:1–43.

BUNN, S. E., AND P. M. DAVIES. 2000. Biological processes in running waters and their implications for the assessment of ecological integrity. Hydrobiologia 422:61–70.

CASAS, J. J., AND M. O. GESSNER. 1999. Leaf litter breakdown in a Mediterranean stream characterized by travertine precipitation. Freshwater Biology 41:781–793.

CHADWICK, M. A., AND A. D. HURYN. 2007. Role of habitat in determining macroinvertebrate production in an intermittent-stream system. Freshwater Biology 52:240–251.

CHAMBLIN, H. D., P. B. WOOD, AND J. W. EDWARDS. 2004. Allegheny woodrat (*Neotoma magister*) use of rock drainage channels on reclaimed mines in southern West Virginia. American Midland Naturalist 151:346–354.

CLARKE, K. R., AND R. M. WARWICK. 2001. Change in marine communities: an approach to statistical analysis and interpretation. 2nd edition. PRIMER-E, Plymouth, UK.

CRAW, D., C. G. RUFAUT, S. HAMMIT, S. G. CLEARWATER, AND C. M. SMITH. 2007. Geological controls on natural ecosystem recovery on mine waste in southern New Zealand. Environmental Geology 51:1389–1400.

DAVIES, S. P., AND S. K. JACKSON. 2006. The biological condition gradient: a descriptive model for interpreting change in aquatic ecosystems. Ecological Applications 16:1251–1266.

DOW, C. L., AND R. A. ZAMPELLA. 2000. Specific conductance and pH as indicators of watershed disturbance in streams of the New Jersey Pinelands, USA. Environmental Management 26:437–446.

DUFFY, P. A. 2003. How filled was my valley: continuing the debate on disposal impacts. Natural Resources and Environment 17:143–145, 177–180.

EDWARDS, C. A. 1967. Relationships between weights, volumes and numbers of soil animals. Pages 585–594 *in* O. Graff and J. E. Satchell (editors). Progress in soil biology. North-Holland Publishing Company, Amsterdam, The Netherlands.

FIELD, J. G., K. R. CLARKE, AND R. M. WARWICK. 1982. A practical strategy for analyzing multispecies distribution patterns. Marine Ecology Progress Series 8:37–52.

FOX, J. 1999. Mountaintop removal in West Virginia: an environmental sacrifice zone. Organization and Environment 12:163–183.

FRITZ, K. M., B. R. JOHNSON, AND D. M. WALTERS. 2006. Field operations manual for assessing the hydrologic permanence and ecological condition of headwater streams. EPA 600/R-06/126. Office of Research and Development, National Exposure Research Laboratory, US Environmental Protection Agency, Cincinnati, Ohio. (Available from: http://www.epa.gov/eerd/manual/headwater.htm)

GESSNER, M. O., AND E. CHAUVET. 2002. A case for using litter breakdown to assess functional stream integrity. Ecological Applications 12:498–510.

GHIORSE, W. C. 1984. Biology of iron- and manganese-depositing bacteria. Annual Review of Microbiology 38: 515–550.

GRAY, L. J., AND J. V. WARD. 1983. Leaf litter breakdown in streams receiving treated and untreated metal mine drainage. Environmental International 9:135–138.

GREENE, B. C., AND W. B. RANEY. 1979. West Virginia's controlled placement method of surface mining: maximum recovery – minimum disturbance. Pages 438–453 *in* C. O. Brawner and I. P. F. Dorling (editors). Stability in coal mining. Proceedings of the 1st International Symposium on Stability in Coal Mining. Miller Freeman Publications, San Francisco, California.

HARTMAN, K. J., M. D. KALLER, J. W. HOWELL, AND J. A. SWEKA. 2005. How much do valley fills influence headwater streams? Hydrobiologia 532:91–102.

HAYES, R. A. 1998. Soil survey of Breathitt County, Kentucky. Natural Resources Conservation Service, US Department of Agriculture, Washington, DC. (Available from: State Conservationist, 771 Corporate Drive, Suite 210, Lexington, Kentucky 40503 USA)

HOLL, K. D. 2002. Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. Journal of Applied Ecology 39:960–970.

HOOKE, R. L. 1999. Spatial distribution of human geomorphic activity in the United States: comparison with rivers. Earth Surface Processes and Landforms 24:687–692.

HURYN, A. D., V. M. BUTZ HURYN, C. J. ARBUCKLE, AND L. TSOMIDES. 2002. Catchment land-use, macroinvertebrates and detritus processing in headwater streams: taxonomic richness versus function. Freshwater Biology 47: 401–415.

JACK, J. D., A. C. PAROLA, W. S. VESELY, M. A. CROASDAILE, AND R. H. KELLEY. 2005. Evaluation and assessment of stream restoration projects: lessons learned and integration of a watershed perspective. Final report to Kentucky Division of Water, Frankfort, Kentucky. (Available from: Kentucky Division of Water, 200 Fair Oaks Lane, Frankfort, Kentucky 40601 USA)

KELLY, M. 1988. Mining and the freshwater environment. Elsevier Applied Science, New York.

KENTUCKY GEOLOGICAL SURVEY. 2009. Coal production database. Kentucky Geological Survey, Lexington, Kentucky. (Available from: http://kgs.uky.edu/kgsweb/DataSearching/Coal/Production/prodsearch.asp)

LARNED, S. T., T. DATRY, AND C. T. ROBINSON. 2007. Invertebrate and microbial responses to inundation in an ephemeral river reach in New Zealand: effects of preceding dry periods. Aquatic Science 69:554–567.

MATTHEWS, R. A., A. L. BUIKEMA, J. CAIRNS, AND J. H. RODGERS. 1982. Biological monitoring Part IIA – receiving system functional methods, relationships and indices. Water Research 16:129–139.

MERRICKS, T. C., D. S. CHERRY, C. E. ZIPPER, R. J. CURRIE, AND T. W. VALENTI. 2007. Coal-mine hollow fill and settling pond influences on headwater streams in southern West Virginia, USA. Environmental Monitoring and Assessment 129:359–378.

MESSINGER, T., AND K. S. PAYBINS. 2003. Relations between precipitation and daily and monthly mean flows in gaged, unmined and valley-filled watersheds, Ballard Fork, West Virginia, 1999–2001. U.S. Geological Survey Water-Resources Investigations Report 03-4113. US Geological Survey, Charleston, West Virginia.

MEYER, J. L. 1997. Stream health: incorporating the human dimension to advance stream ecology. Journal of the North American Benthological Society 16:439–447.

MONTGOMERY, H. J., C. M. MONREAL, J. C. YOUNG, AND K. A. SEIFERT. 2000. Determination of soil fungal biomass from soil ergosterol analyses. Soil Biology and Biochemistry 32:1207–1217.

NIYOGI, D. K., W. M. MATTHEWS, AND D. H. MCKNIGHT. 2001. Litter breakdown in mountain streams affected by mine drainage: biotic mediation of abiotic controls. Ecological Applications 11:506–516.

NIYOGI, D. K., D. H. MCKNIGHT, AND W. M. MATTHEWS. 2002. Effects of mine drainage on breakdown of aspen litter in mountain streams. Water, Air, and Soil Pollution: Focus 2:329–341.

OWENBY, J., R. HEIM, M. BURGIN, AND D. EZELL. 2001. Climatography of the U.S. No. 81 – Supplement #3. Maps of annual 1961–1990 normal temperature, precipitation and degree days. National Climate Data Center, Asheville, North Carolina. (Available from: http://lwf.ncdc.noaa.gov/oa/documentlibrary/clim81supp3/clim81.html)

PAUL, M. J., AND J. L. MEYER. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

PHILLIPPI, M. A., AND A. BOEBINGER. 1986. A vegetational analysis of three small watersheds in Robinson Forest, Eastern Kentucky. Castanea 51:11–30.

PHILLIPS, J. D. 2004. Impacts of surface mine valley fills on headwater floods in eastern Kentucky. Environmental Geology 45:367–380.

POND, G. J., AND S. E. MCMURRAY. 2002. A macroinvertebrate bioassessment index for headwater streams of the Eastern Coalfield Region, Kentucky. Kentucky Department for Environmental Protection, Frankfort, Kentucky. (Available from: http://www.water.ky.gov/NR/rdonlyres/1F744F9F-8C03-4E11-BF0D-55CAE23E3969/0/EKyMBI1.pdf)

POND, G. J., M. E. PASSMORE, F. A. BORSUK, L. REYNOLDS, AND C. A. ROSE. 2008. Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. Journal of the North American Benthological Society 27: 717–737.

ROBINS, J. D. 1979. Constructing spoil storage fills in the steep terrain Appalachian Coalfields. Pages 463–470 *in* C. O. Brawner and I. P. F. Dorling (editors). Stability in coal mining. Proceedings of the 1st International Symposium on Stability in Coal Mining. Miller Freeman Publications, San Francisco, California.

ROSSET, J., F. BÄRLOCHER, AND J. J. OERTLI. 1982. Decomposition of conifer needles and deciduous leaves in two Black Forest and two Swiss Jura streams. Internationale Revue der gesamten Hydrobiologie 67:695–711.

ROYER, T. V., AND G. W. MINSHALL. 2003. Controls on leaf processing in streams from spatial-scaling and hierarchical perspectives. Journal of the North American Benthological Society 22:352–358.

SAMPLE, B. E., R. J. COOPER, R. D. GREER, AND R. C. WHITMORE. 1993. Estimation of insect biomass by length and width. American Midland Naturalist 129:234–240.

SCHINDLER, D. W. 1987. Detecting ecosystem responses to anthropogenic stress. Canadian Journal of Fisheries and Aquatic Sciences Supplement 44:6–25.

SCHLIEF, J. 2004. Leaf associated microbial activities in a stream affected by acid mine drainage. International Review of Hydrobiology 89:467–475.

SIEFERT, J., AND M. MUTZ. 2001. Processing of leaf litter in acid waters of the post-mining landscape in Lusatia, Germany. Ecological Engineering 17:297–306.

SIMMONS, J. A., W. S. CURRIE, K. N. ESHLEMAN, K. KUERS, S. MONTELEONE, T. L. NEGLEY, B. R. POHLAD, AND C. L. THOMAS. 2008. Forest to reclaimed mine land use change leads to altered ecosystem structure and function. Ecological Applications 18:104–118.

SKOUSEN, J., P. ZIEMKIEWICZ, AND C. VENABLE. 2006. Tree recruitment and growth on 20-year-old, unreclaimed surface mined lands in West Virginia. International Journal of Mining, Reclamation and Environment 20: 142–154.

SLONECKER, E. T., AND M. J. BENGER. 2002. Remote sensing and mountaintop mining. Remote Sensing Reviews 20: 293–322.

SPARKS, J., T. HAGMAN, D. MESSER, AND J. TOWNSEND. 2003a. Eastern Kentucky stream assessment protocol: utility in making mitigation decisions. Aquatic Resources News: A Regulatory Newsletter 2(2):4–10. (Available from: http://140.194.76.129/cw/cecwo/reg/aqua/vol2-01.pdf)

SPARKS, J., J. TOWNSEND, T. HAGMAN, AND D. MESSER. 2003b. Stream assessment protocol for headwater streams in the Eastern Kentucky Coalfield Region. Aquatic Resources News: A Regulatory Newsletter 2(1):2–5. (Available from: http://140.194.76.129/cw/cecwo/reg/aqua/vol2-2.pdf)

SZWILSKI, T. B., B. E. DULIN, AND J. W. HOOPER. 2001. An innovative approach to managing the environmental impacts of mountaintop coal mining in West Virginia. International Journal of Surface Mining, Reclamation, and Environment 15:73–85.

TUCH, K., AND A. GASITH. 1989. Effects of an upland impoundment on structural and functional properties of a small stream in a basaltic plateau (Golan Heights, Israel). Regulated Rivers: Research and Management 3: 153–167.

USDOE-EIA (US DEPARTMENT OF ENERGY, ENERGY INFORMATION ADMINISTRATION). 2008a. Annual coal report, 2007. Energy Information Administration, US Department of Energy, Washington, DC. (Available from: http://www.eia.doe.gov/cneaf/coal/page/acr/acr_sum.html)

USDOE-EIA (US DEPARTMENT OF ENERGY, ENERGY INFORMATION ADMINISTRATION). 2008b. Annual energy outlook report, 2008. Energy Information Administration, US Department of Energy, Washington, DC. (Available from: http://www.eia.doe.gov/oiaf/aeo/excel/aeotab_15.xls)

USDOE-EIA (US DEPARTMENT OF ENERGY, ENERGY INFORMATION ADMINISTRATION). 2008c. International energy annual report, 2008. Energy Information Administration, US Department of Energy, Washington, DC. (Available from: http://www.eia.doe.gov/emeu/international/RecentPrimaryCoalProductionMST.xls)

USEPA (US ENVIRONMENTAL PROTECTION AGENCY). 2005. Mountaintop mining/valley fills in Appalachia. Final programmatic environmental impact statement. Region 3, US Environmental Protection Agency, Philadelphia, Pennsylvania. (Available from: http://www.epa.gov/region3/mtntop/index.htm)

VUORI, K.-M. 1995. Direct and indirect effects of iron on river ecosystems. Annales Zoologici Fennici 32:317–329.

WALLACE, J. B., J. W. GRUBAUGH, AND M. R. WHILES. 1996. Biotic indices and stream ecosystem processes: results from an experimental study. Ecological Applications 6: 140–151.

WANG, Z., AND L. A. GOONEWARDENE. 2004. The use of MIXED models in the analysis of animal experiments with repeated measures data. Canadian Journal of Animal Science 84:1–11.

WICKHAM, J. D., K. H. RIITTERS, T. G. WADE, M. COAN, AND C. HOMER. 2007. The effect of Appalachian mountaintop mining on interior forest. Landscape Ecology 22: 179–187.

WILEY, J. B., R. D. EVALDI, J. H. EYCHANER, AND D. B. CHAMBERS. 2001. Reconnaissance of stream geomorphology, low streamflow, and stream temperature in the mountaintop coal-mining region, southern West Virginia, 1999–2000. U.S. Geological Survey Water-Resources Investigations Report 01-4092. US Geological Survey, Charleston, West Virginia.

WOOD, P. B., S. B. BOSWORTH, AND R. DETTMERS. 2006. Cerulean warbler abundance and occurrence relative to large-scale edge and habitat characteristics. Condor 108: 154–165.

WOODS, A. J., J. M. OMERNIK, W. H. MARTIN, G. J. POND, W. M. ANDREWS, S. M. CALL, J. A. COMSTOCK, AND D. D. TAYLOR. 2002. Ecoregions of Kentucky (2 sided color poster with map, descriptive text, summary tables, and photographs) (map scale 1:1,000,000). US Geological Survey, Reston, Virginia. (Available from: ftp://ftp.epa.gov/wed/ecoregions/ky/ky_eco_lg.pdf)

YOUNG, R. G., C. D. MATTHAEI, AND C. R. TOWNSEND. 2008. Organic matter breakdown and ecosystem metabolism: functional indicators for assessing river ecosystem health. Journal of the North American Benthological Society 27:605–625.

Received: 13 May 2009
Accepted: 19 February 2010